AN ABSTRACT OF THE THESIS OF

Heather E. Greaves for the degree of Master of Science in Forest Science presented on October 17, 2012.

Title: Potential Effects of Climate Change and Fire Management on Fire Behavior and Vegetation Patterns on an East Cascades Landscape

Abstract approved:

Brenda C. McComb

Climate exerts considerable control on wildfire regimes, and climate and wildfire are both major drivers of forest growth and succession in interior Northwest forests. Estimating potential response of these landscapes to anticipated changes in climate helps researchers and land managers understand and mitigate impacts of climate change on important ecological and economic resources. Spatially explicit, mechanistic computer simulation models are powerful tools that permit researchers to incorporate climate and disturbance events along with vegetation physiology and phenology to explore complex potential effects of climate change over wide spatial and temporal scales. In this thesis, I used the simulation model FireBGCv2 to characterize potential response of fire, vegetation, and landscape dynamics to a range of possible future climate and fire management scenarios. The simulation landscape (~43,000 hectares) is part of Deschutes National Forest, which is located at the interface of maritime and continental climates and is known for its beauty and ecological diversity. Simulation scenarios included all combinations of +0°C, +3°C, and +6°C of warming; +10%, ±0%, and -10% historical precipitation; and 10% and 90% fire suppression, and were run for 500 years. To characterize fire dynamics, I investigated how mean fire frequency, intensity, and fuel loadings changed over time in all scenarios, and how fire and tree mortality interacted over time. To explore vegetation and landscape dynamics, I described the distribution and spatial arrangement of vegetation types and forest successional stages on the landscape, and used a nonmetric multidimensional

scaling (NMS) ordination to holistically evaluate overall similarity of composition, structure, and landscape pattern among all simulation scenarios over time.

Changes in precipitation had little effect on fire characteristics or vegetation and landscape characteristics, indicating that simulated precipitation changes were not sufficient to significantly affect vegetation moisture stress or fire behavior on this landscape. Current heavy fuel loads controlled early fire dynamics, with high mean fire intensities occurring early in all simulations. Increases in fire frequency accompanied all temperature increases, leading to decreasing fuel loads and fire intensities over time in warming scenarios. With no increase in temperature or in fire frequency, high fire intensities and heavier fuel loads were sustained. Over time, more fire associated with warming or less fire suppression increased the percentage of the landscape occupied by non-forest and fire-sensitive early seral forest successional stages, which tended to increase the percentage of fire area burning at high severity (in terms of tree mortality). This fire-vegetation relationship may reflect a return to a more historical range of conditions on this landscape.

Higher temperatures and fire frequency led to significant spatial migration of forest types across the landscape, with communities at the highest and lowest elevations particularly affected. Warming led to an upslope shift of warm mixed conifer and ponderosa pine (*Pinus ponderosa*) forests, severely contracting (under 3° of warming) or eliminating (under 6° of warming) area dominated by mountain hemlock (*Tsuga mertensiana*) and cool, wet conifer forest in the high western portion of the landscape. In lower elevations, warming and fire together contributed to significant expansion of open (<10% tree canopy cover) forest and grass- and shrubland. The compositional changes and spatial shifts simulated in the warming scenarios suggest that climate change is likely to significantly affect forests on this landscape. Warming and associated fire also tended to increase heterogeneity of forest structural stages and landscape pattern, resulting in a more diverse distribution of structural stages, especially in lower elevations, and a more divided landscape of smaller forest stands.

The NMS ordination emphasized the dissimilarity between the severe +6° scenarios and the other two temperature scenarios. The +0° and +3° scenarios differed from each other in composition (mainly because cool forest was lost in the +3° scenarios), but within a given level of fire suppression they remained remarkably similar in terms of overall composition, structure, and landscape pattern, while the +6° scenarios separated noticeably from them. Such decisive differences suggest that under the simulated ranges of precipitation and fire suppression, the interval between 3 and 6 degrees of warming on this landscape may capture an ecological threshold, or tipping point.

Additional simulation research that incorporates (for example) management actions, insects and pathogens, and a wider array of precipitation scenarios could help illuminate more clearly the possible range of future landscape conditions. Still, these results provide a glimpse of potential divergent outcomes on this important landscape under possible future climates, and suggest that these forests will undergo considerable changes from both historical and current conditions in response to higher temperatures expected in this area. Some changes may be inevitable with warming, such as the upslope shift of warm forest types, but careful planning for fire and fuels management might allow land managers to modulate fire behavior and steer vegetation dynamics toward the most desirable outcome possible.

Potential Effects of Climate Change and Fire Management on Fire Behavior and Vegetation Patterns on an East Cascades Landscape

by
Heather E. Greaves

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Master of Science

Presented October 17, 2012
Commencement June 2013

Master of Science thesis of Heather E. Greaves presented on October 17, 2012.

APPROVED:

Major Professor, representing Forest Science

Head of the Department of Forest Ecosystems and Society

Dean of the Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Heather E. Greaves, Author

ACKNOWLEDGEMENTS

This research is part of the Integrated Landscape Assessment Project, funded by the USDA Forest Service PNW Research Station under the American Recovery and Reinvestment Act. I was also supported by an Oregon Lottery Scholarship, which was greatly appreciated. Additional funding came from Joint Fire Science Program project 09-1-08-31, thanks to John Lehmkuhl at the USDA Forest Service. Statistical, computing, and logistic support were provided by the Department of Forest Ecosystems and Society, Oregon State University, and the USDA Forest Service PNW Research Station, Corvallis, Oregon.

Numerous people helped me on my way to completing this work, and I am grateful to all of them. Rebecca Kennedy brought me onto this project, and provided direction until she departed to pursue new opportunities. Brenda McComb gracefully stepped in to advise me, and I am deeply thankful for her sympathetic guidance, level-headed pragmatism, and dry sense of humor. Bob Keane fielded endless questions about FireBGCv2 with patience and understanding, was kind enough to join my committee, and provided perceptive feedback on thesis drafts. John Bailey, Oregon State's fire ecology evangelist, also provided valuable feedback as a committee member, and presided over numerous classes and extracurricular discussions that influenced my thinking about this work. Michael Campana graciously assented to join my committee in the capacity of Graduate Council Representative.

At various times, Heather Roberts, Joe Bernert, Roger Ottmar, Jason Clark, Lisa Holsinger, Dave Shaw, Travis Woolley, Josh Halofsky, Treg Christopher, Janine Salwasser and numerous others gave sorely needed help with data acquisition, scripting, calculations, or advice on parameterization and processing. Becky Kerns braved deep bureaucracy to ensure I retained an office. Erik Haunreiter gamely assisted me through my early GIS ignorance, and subsequently provided occasional advice and invaluable camaraderie. Patrick Fekety and the OSU Pyromaniacs—including associated friends, musicians, and benevolent nutcases—formed the bedrock of my sanity, and deserve more thanks than I can express. I hope to have the privilege of working with many Pyromaniacs in the future.

Finally, my parents have shown remarkable patience and equanimity toward my wending path through life, and provide continuous, unobtrusive support and inspiration. I am enduringly grateful for everything they have done for me.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDIX FIGURES

LIST OF APPENDIX TABLES

POTENTIAL EFFECTS OF CLIMATE CHANGE AND FIRE MANAGEMENT ON FIRE BEHAVIOR AND VEGETATION PATTERNS ON AN EAST CASCADES LANDSCAPE

CHAPTER 1—INTRODUCTION

In the dry interior forests of the Pacific Northwest, the past century has been a time of rapid ecological change, a trend the next century promises to prolong. Since widespread settlement by Euroamericans in the early 1900s, logging, grazing, and the introduction of non-native plant species have changed the face of the landscape, while concerted efforts to eliminate perceived destructive effects of wildfire have redirected structural and successional pathways by which these forests historically grew and changed (Hessburg and Agee 2003). Consequently, modern dry forests are markedly different in appearance and function from those encountered by early Euroamericans (Hessburg and Agee 2003, Hessburg et al. 2005, Naficy et al. 2010). But even as researchers, land managers, and policy makers begin to understand how forests are responding to recent changes, warming temperatures associated with global climate change are likely to further alter the composition and function of these systems (Chmura et al. 2011, Rogers et al. 2011, Waring et al. 2011).

The magnitude of temperature changes in the Pacific Northwest over the next century will control the extent to which ecosystems change in response. Much of the anticipated warming is expected to occur during summer months—the primary growing season—with lesser increases in other seasons (Mote and Salathé 2010). By 2080, summer temperatures may be 4.5°C warmer than 1970-1999 means, with some models projecting summer increases up to 6° or 7°C; precipitation changes are more uncertain, though they may be less dramatic than temperature changes (Mote and Salathé 2010).

Climate directly influences phenology and physiology of forest vegetation, and also acts as an important control on wildfire, which was historically the major disturbance vector in dry forests (Heyerdahl et al. 2001, Westerling et al. 2006, Littell et al. 2009). Warming temperatures are likely to geographically shift environmental envelopes to which all vegetation types are adapted, forcing vegetation to shift as well (Gonzalez et al. 2010) and leading to new arrangements of species and forest types on the landscape. Systems that already occupy an environmental extreme, such as subalpine forest, may contract if they have nowhere to go

(Busing et al. 2007, Lenihan et al. 2008b). Meanwhile, longer and warmer fire seasons already appear to provide more opportunities for fire ignition, increasing fire frequency and annual average area burned (Westerling et al. 2006, Littell et al. 2009). Heavy fuels accumulated during fire exclusion tend to make these fires more intense, leading to unusually large areas of extensive tree mortality (USFS 2004, 2005, Miller et al. 2009). Along with the direct effect of warming, these more frequent, intense fires have the potential to further alter these already-changed landscapes.

Humans also have a large impact on East Cascades fire regimes, especially since the advent of fire exclusion efforts in the early- and mid-1900s (Everett et al. 2000, Hessburg and Agee 2003). The ecological necessity of fire in these systems has recently become better understood, and land managers have sought to re-introduce fire as a tool for influencing vegetation structure and behavior of subsequent wildfires (Graham et al. 2004), but the dangers that wildfire poses to ecological and economic values ensure that wildfire suppression will remain a priority. The degree to which wildfires are prevented from burning in the future will likely have important consequences for the composition and structure of vegetation, as well as the impacts of fires that escape suppression (Lenihan et al. 2008b).

Climate, vegetation, and fire regime interact in complicated ways at multiple spatial and temporal scales, making the effects of climate change on landscapes difficult to project into the future. Computer simulation modeling of landscape ecological processes is a growing field of inquiry that addresses these challenges by allowing researchers to investigate complex ecosystem responses to multiple possible futures (Keane et al. 2004, He et al. 2008), providing useful glimpses of potential paths that landscapes may follow under a range of conditions.

I used FireBGCv2, a multiscale spatial model that simulates fire disturbance and vegetation regeneration, growth, and mortality under scenarios with altered temperature, atmospheric carbon dioxide (CO_2) concentration, precipitation, and fire suppression. Climate change values encapsulate changes anticipated by Mote and Salathé (2010) under different carbon emissions scenarios, downscaled for the Northwest from a suite of global circulation models. The simulation area is a landscape in Deschutes National Forest, Oregon, on the eastern slopes of the Cascade Range. Deschutes National Forest is a large, popular recreation area that is estimated to receive more than 8 million visitors per year (USFS 2012), who take advantage of opportunities for hiking, camping, mountain biking, hunting, fishing, and skiing in a landscape known for its volcanic beauty and ecological diversity. Simulations of the potential effects of climate change on this important area provide insight into future vegetation and fire dynamics that may have large ecological and economic impacts. They also contribute to the growing body of knowledge regarding projected future vegetation characteristics and fire effects in similar dry interior Northwest forests, which may prove helpful to researchers and managers planning for future forest wildlife habitat, resource use, and other ecosystem services.

Chapter 2 of this thesis describes simulation results related to fire dynamics. Specifically, I describe how the percentage of area burning at high severity varied with changes in temperature, precipitation, and fire suppression levels, and how that relates to patterns of fire frequency and fire intensity, examining trends over time as well as overall averages. I also examine the interaction between fire dynamics and vegetation characteristics, because the two are inherently entwined but may respond divergently to changes in future climate.

In Chapter 3, I explore the simulated response of vegetation and landscape attributes to future climates and fire suppression levels. The current landscape encompasses a wide range of forest types, from hot, flat, low-elevation pine forests to high, cold, subalpine hemlock (*Tsuga*) and fir (*Abies*), which may respond differently under future climates. In particular, I address the following questions: First, how might warming climate and different fire suppression levels affect overall landscape composition, structure, and configuration over time? And second, how might distributions of major vegetation types differ among these potential future scenarios after 500 years? I also describe potential changes in forest structure and landscape configuration, which have particular relevance to wildlife habitat and efforts to restore forests to pre-Euroamerican conditions.

Chapter 4 concludes this thesis, summarizing themes from the previous chapters and exploring the wider implications of the findings.

CHAPTER 2—FIRE DYNAMICS

ABSTRACT

In the Oregon East Cascades, any changes in vegetation and disturbance regimes due to climate warming will impact wildlife habitat, timber and water resources, and recreational opportunities on both public and private lands. However, complicated interactions among physiology, phenology, and disturbance are difficult for researchers to quantify and synthesize, especially at the large spatial and temporal scales involved. Recent fires in the Deschutes National Forest have been uncharacteristically large and intense, and anticipated climate warming may exacerbate changes to a fire regime already altered by 20th-century Euroamerican land and fire management.

I used FireBGCv2, a spatially explicit mechanistic forest succession and disturbance model, to simulate fire, vegetation, and landscape dynamics in a portion of the Deschutes National Forest under 18 potential climate change scenarios. In particular, I investigated how mean fire intensity and the percentage of fire area that burned severely responded to potential climate and fire management, and how these fire dynamics interacted with fuel loads and vegetation. Simulation scenarios included all combinations of +0°C, +3°C, and +6°C of warming; +10%, ±0%, and -10% historical precipitation; and 10% and 90% fire suppression, and were run for 500 years.

Changes in precipitation did not strongly affect fire dynamics. Averaged over time, rising temperatures decreased mean fire intensity, as more-frequent fire reduced fuel loads and fire intensities sooner. However, because warming and frequent fire increased the prominence of non-forest and young forest, which is vulnerable to fire, the proportion of high-severity burned area increased with temperature. In scenarios with less fire suppression, this trend was amplified. In the short term, warming destabilized the fire regime on this landscape, and fire suppression promoted high-intensity fires, delaying the establishment of a new, more stable fire regime.

INTRODUCTION

Because temperature and precipitation are major drivers of wildfire dynamics, anticipated changes in global climate are likely to influence the behavior and impact of future fire in fire-dependent ecosystems worldwide (Bowman et al. 2009). Climate-related changes in dry forests will manifest directly, by altering species distribution and migration (Coops and Waring 2011), and indirectly, via altered wildfire regimes (wildfire size, frequency, and severity), as well as through vegetation-fire interactions (Westerling et al. 2003, McKenzie et al. 2004, Haugo et al. 2010, Liu et al. 2010). Wildfire was historically the primary disturbance in forests of the interior Northwest (Agee 1994, Hessburg and Agee 2003), and after nearly a century of fire suppression, climate change and land management activities altering forest structure may be spurring a resurgence in fire's importance in this region (Westerling et al. 2006, Littell et al. 2009, Miller et al. 2009). However, both the magnitude of climate change and how it may interact with fire management strategies remain uncertain; these interactions depend on highly localized variables, making them difficult to predict. Computer simulation models, which can project the effects of climate and management scenarios on landscapes over long temporal scales, have emerged as valuable tools to address these uncertainties and explore potential trends in fire and vegetation dynamics.

Numerous studies examine the relationship between climate and wildfire, and foresee increases in wildfire frequency and size in response to anticipated climate warming (e.g., Heyerdahl et al. 2001, Westerling et al. 2006, Littell et al. 2009), but relatively few investigate future fire intensity (energy released per unit area) or fire severity (ecological effect of fire) which are more difficult to project but are arguably the principal determinants of fires' impact (Flannigan et al. 2009, Littell et al. 2009, Hessl 2011). Fire intensity is an especially important aspect of fire behavior, because high-intensity fires are more difficult to control and extinguish, and are more likely to burn at high severity with significant consequences for values like wildlife habitat, timber reserves, and carbon storage (Ager et al. 2007, Flannigan et al. 2009, Meigs et al. 2009). Fire intensity is determined at fine scales by live and dead fuel loadings and by weather conditions, all of which may be affected by changes in climate (Flannigan et al. 2009). Estimates of possible trends in future fire intensities would help land managers plan fuel treatments and fire suppression strategies in these fire-prone Western forests.

Fire severity refers generally to the impact of fire on an ecosystem, but precise definitions vary in the literature and are often conflated with other fire descriptors (reviewed in Keeley 2009), making comparison across studies difficult. Depending on the focus of a given study, “fire severity” might refer to biomass consumed by fire, extent of soil heating, total fire carbon release, change in landscape greenness, extent of tree mortality (as in this study), or extent of mortality of large trees only. Under most definitions, though, fire severity emerges from complex interplay between fire behavior and vegetation that is difficult to predict or model over large temporal or spatial scales.

When severity is defined as the extent of tree mortality, the proportion of fire area burning at high severity in the West appears to be increasing, a trend that is probably partly due to fuel loads accumulated during 20^{th}-century fire exclusion efforts (Hessburg et al. 2005, Dillon et al. 2011) and partly due to climate (Miller et al. 2009). In Deschutes National Forest in 2003, for example, the Davis and B&B Complex fires burned extensive areas of mixed-conifer forest, with heavy surface and canopy fuel accumulations contributing to uncharacteristically large percentages of fire area with near-total tree mortality (80% and 46% of total fire area for Davis and B&B fires, respectively; USFS 2004, 2005). These forests, adapted to a more high-frequency, low- or mixed-severity fire regime, were far outside their historical fire return intervals (Hessburg et al. 2005). By modifying fire regimes, it appears that humans have also modified trends in fire intensity and severity, which may in turn alter the vegetation and structure of the forests themselves (Savage and Mast 2005, Westerling et al. 2011), while increasing fire carbon release (Meigs et al. 2009) and destroying forests set aside for wildlife habitat (USFS 2004, 2005).

Whereas heavier fuel loads represent an anthropogenic alteration to bottom-up controls on fire, changes in climate represent alterations to fire’s main top-down control (Heyerdahl et al. 2002). Fuels that are continually and increasingly dry due to prolonged higher temperatures will tend to increase fire intensities, and consequently increase high-severity burned area, an effect that may already be occurring in the West (Lutz et al. 2009, Miller et al. 2009), and may be amplified if fuels also become denser and more continuous as a result of climate-induced increases in vegetation productivity (Holden et al. 2007, Hessl 2011, Rogers et al. 2011). The complex interactions among fuels, vegetation, and fire behavior preclude simple conclusions. In some areas, for example, fire intensities may decrease where fuels become lighter and less continuous as a result of heat and drought stress (Dillon et al. 2011, Hessl 2011, Miller et al. 2012).

Future fire behavior will also respond to fuel and vegetation conditions determined by human fire and fuels management (Lenihan et al. 2008b). For the majority of the last century, fire policies focused exclusively on eliminating fire from Western landscapes to preserve timber and water resources (Hessburg and Agee 2003). Recently, however, land managers have begun to recognize the integral role that fire plays in these ecosystems, and have sought to reintroduce fire as a tool to reduce fuel loads and control vegetation structure and the behavior of subsequent wildfire (Graham et al. 2004). Allowing wildfires to burn ("wildland fire use") whenever feasible is increasingly seen as a reasonable and cost-effective fire management approach (van Wagtendonk 2007), but determining how future fire management strategies and potential climate changes may affect focal landscapes is a challenge for researchers and land managers trying to prepare for an uncertain future.

In the East Cascades, it is widely accepted that although fire was historically common, the majority of low- and mid-elevation burned area did not experience high tree mortality, but recent fire events raise questions about the persistence of high-frequency, low- and mixed-severity fire regimes, and about the potential successional trajectories of forests adapted to them. Given the challenges of predicting the complicated relationships involved, simulation models provide a valuable tool for researchers seeking to explore potential outcomes of future climate and management scenarios on fire dynamics in specific landscapes. I used a spatially explicit, individual-tree-based simulation model to explore these questions on a landscape in Oregon's East Cascades. Specifically, my objective was to characterize and compare fire intensity and severity over time in scenarios of potential future temperature, precipitation, and fire suppression levels.

METHODS

Study area

The simulation landscape represents ~43,000 hectares in the Cascade Lakes area of Deschutes National Forest, along the eastern slopes of the Cascade Mountains, and captures the dynamic transition zone between wet maritime and dry interior climate zones (Figure 1). Elevations range from roughly 1300 meters in the east to over 2700 meters at the volcanic peaks to the north and west. This region is characterized by broad plains sloping shallowly from the Cascade crest down to eastern flats, punctuated by cinder cones (Franklin and Dyrness 1988). Mean annual precipitation ranges from 260 to 2800 mm, falling mostly as winter snow; mean annual minimum and maximum temperatures vary from -15° to 30° C (PRISM 2011). Soils in the region are young volcanic, with patches of lava flows and deposits of ash and pumice, and thus tend to be shallow or have poor water-holding capacity (Simpson 2007).

The relatively simple topography and gradients of elevation, temperature, and precipitation give rise to distinct bands of dominant forest vegetation types (Franklin and Dyrness 1988, Simpson 2007). At higher elevations, mountain hemlock (*Tsuga mertensiana*) and Pacific silver fir (*Abies amabilis*) dominate, with cool mixed conifer forests of lodgepole pine (*Pinus contorta*), western white pine (*Pinus monticola*), subalpine fir (*Abies lasiocarpa*), Shasta red fir (*Abies magnifica shastensis*), and whitebark pine (*Pinus albicaulis*) also present. Moist mixed conifer forests may occur below the upper montane zone, and include grand fir/white fir (*Abies grandis/concolor*), Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), Shasta red fir, and Engelmann spruce (*Picea engelmannii*). On drier sites and southern, warmer aspects, dry mixed conifer forests dominate, characterized by ponderosa pine, Douglas-fir, lodgepole pine, incense-cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and grand fir/white fir. Vegetation on cooler northern aspects may resemble that on upper montane sites. As the landscape grades into eastern flats, large stands of lodgepole pine occupy cold dry pockets or old burn perimeters; on warmer flats, lodgepole pine intergrades with ponderosa pine or gives way to stands of pure ponderosa pine. Shrublands dominated by big sagebrush associations (*Artemisia tridentata*) and western juniper (*Juniperus occidentalis*) are found in the drier, hotter east. In this study, the starting simulation landscape encompasses nearly the full range of these forest types, excluding only the juniper-big sagebrush associations.

Prior to the advent of fire exclusion efforts in the mid 1900s, fire return intervals in the study area ranged from less than ten years in lower elevations to at least 200-300 years in higher montane sites (Bork 1985, Agee 1994, Hessburg and Agee 2003, Wright and Agee 2004).

Fire severity (in terms of overstory mortality) generally corresponded inversely to fire frequency, with frequent low-severity fires in low-elevation dry forests, and rare high-severity fires in the higher, moister forests (but see Baker 2012); mixed-severity fire regimes characterized areas of moderate elevation and moisture (Hessburg et al. 2005, Perry et al. 2011).

Model description

FireBGCv2 (Keane et al. 2011) is a spatially explicit, mechanistic, individual-tree simulation modeling platform with modules for ecosystem processes, including wildfire, vegetation dynamics, weather, and hydrological systems. It operates at multiple spatial scales (landscape, site, plot, species, tree) and two temporal scales (daily and annual), depending on the process being simulated (Figure 2, Table 1). FireBGCv2 and its predecessor, FIRE-BGC, have been used extensively to explore complex fire and vegetation dynamics throughout the western U.S. (e.g. Keane et al. 1996, Loehman et al. 2011, http://www.firelab.org/research-projects/fire-ecology/139-firebgc).

Because wildfire is the most significant disturbance acting on Pacific Northwest landscapes, fire is central in FireBGCv2 and is simulated at all spatial scales. At the landscape level, fire ignition is determined stochastically based on annual climate and user-entered site-level fire return parameters, and fire spread is modeled via cell percolation as in the model LANDSUM (Keane et al. 2006). A 1.5 km buffer was added around the study area to permit realistic fire spread at landscape edges. Fuel moistures for dead fuels are simulated at the site level, and fire behavior is modeled at the stand level using either Rothermel's (1972) or Albini's (1976) equations. Fire-caused tree mortality is calculated at the tree level, according to individual species characteristics including bark thickness and height to live crown. Stand boundaries are allowed to shift within site boundaries as fire alters the arrangement of species and successional stages on the landscape. This cross-scale simulation of fire allows for spatially responsive wildfire behavior and effects, both within sites and across the landscape.

Model parameterization and calibration

FireBGCv2 is a large, complex spatial model, and a wide variety of sources were tapped for parameterization. Input requirements for all spatial scales are described more completely in Keane et al. (2011); major data sources and model parameter values described below can be found in Appendix Tables A1 and A2. Site boundaries were delineated based on elevation and potential vegetation types, which also capture variation in climate and soils sufficiently for model parameterization. GIS layers from the Willamette and Deschutes National Forests were used to characterize soil depth and fractional soil components. Sixty-nine years of historical daily weather data from the nearby Wickiup Dam COOP meteorological station were spatially interpolated across the local topography with the model MTCLIM for use in FireBGCv2's climate and weather processes. Information from SNOTEL and DAYMET was also used to fill in weather gaps, and to calculate average lapse rates for input to MTCLIM.

Stands were delineated based on current vegetation cover generalized to patches of at least 9 ha. Spatially continuous gradient nearest neighbor maps generated at 30 x 30 m resolution by the Landscape Ecology Modeling, Mapping & Analysis group (LEMMA) were the source for current vegetation cover. LEMMA data tables were used to generate tree lists and understory biomass loadings for each stand.

Each tree species and understory functional group in FireBGCv2 is parameterized with dozens of morphological, physiological, and phenological attributes, which are described in Keane et al. (2011). Values for the majority of these attributes were found in literature searches; where data were unavailable, estimations were made based on known characteristics from species in similar functional groups from similar geographic regions.

Stand loadings of litter, duff, 1-, 10-, 100-, and 1000-hour fuels were determined by subjectively matching individual stand characteristics (e.g. elevation, cover type, percent cover, number of snags) to appropriate fuelbeds in the Fuel Characteristic Classification System (FCCS) and extracting the associated fuel loadings. FCCS was also used to estimate understory vegetation information for stands that were classified by LEMMA as non-forested, and which therefore lacked LEMMA vegetation data.

FireBGCv2 was run with a calibration scenario consisting of historical weather streams and fire regimes (i.e. no fire suppression), based on the guidelines in Keane et al. (2011). Critical site and species parameters were adjusted until species growth attributes and stand dynamics were stable (did not trend up or down over simulation time) and site-level fire return intervals were within ~20% of historical levels.

Simulation scenarios

To capture uncertainty in the magnitude of potential climate change, multiple combinations of temperature and precipitation changes were investigated (Table 2). Climate change is simulated in FireBGCv2 by modifying the historical daily weather stream according to scaling parameters. In this study, maximum and minimum daily temperatures were altered differentially for each season, with all seasons averaging to a final annual offset of 0, 3, or 6° C. Similarly, seasonal precipitation was altered as a proportion of historical values using unitless scalars, such that the final annual offset from historical values was either zero (1 x historical), 10% lower (0.90 x historical) or 10% higher (1.10 x historical). Temperature and precipitation changes occurred incrementally over the first 100 years of simulation and then were held constant. Atmospheric carbon dioxide (CO_2) levels were increased incrementally in the same fashion. Values for changes in temperature, CO_2, and precipitation, as well as potential trends in seasonality, were based largely on work by Mote and Salathé (2010), with reference to IPCC reports (2007) and others (e.g. Leung et al. 2004, Littell et al. 2010).

To examine effects of fire suppression, the full set of climate scenarios was simulated twice, once with 90% of fires suppressed (approximating current fire suppression levels), and again with 10% suppression (near-historical fire frequency on the landscape). This represents a 3x3x2 factorial design with 18 unique combinations. Because FireBGCv2 incorporates stochastic elements, each climate/fire combination was replicated ten times to capture variability across and within simulation scenarios. Scenarios were simulated for 500 years to allow sufficient time for successional trends to appear; output was produced at 50-year intervals.

Analysis

"High-severity fire" was defined as fire resulting in >70% tree mortality, where proportion tree mortality is calculated per stand as the sum of the squared diameter at breast height (DBH) of all fire-killed trees divided by the sum of the squared DBH of all trees (thereby weighting larger trees more heavily). For each scenario, I divided the area that burned with >70% tree mortality during 100-year intervals into the total area burned in that time and multiplied the result by 100, giving the percentage of high-severity burned area for each simulation replicate. Summary statistics were calculated in R (version 2.14.1; http://www.R-project.org). To determine whether the relative percentage of high-severity fire increased with warming, I divided the total area burned at high severity over the entire simulation by the total area burned, and used simple linear regression to estimate mean overall percentage high-severity burned area and 95% confidence intervals for the mean.

Focusing on the percentage of high-severity burned area separates the ecological effect of fires on the landscape from the frequency that fires occur, but fire frequency, intensity, and severity are inherently intertwined with each other and with the condition of vegetation on the landscape. In FireBGCv2, historical mean fire return intervals for each broad ecophysiological site are set by the user, and then modified by the model based on the fire suppression level for a given scenario. Although the general fire frequency for a given run is therefore set rather than simulated, fire frequency still varies based on annual climate and model stochasticity. Means and standard deviation were therefore calculated for fire rotation (length of time required to burn an area equal to the landscape size), number of simulation years in which fires occurred (fire years), and cumulative area burned over the simulation. It should be noted that variability described by standard deviation reflects variability in model behavior for the given simulation landscape and input parameters, rather than natural ecological variation. Area-weighted fire intensity averaged over 100-year periods was also calculated to provide insight into fire behavior over time, and regressed in R to estimate mean and 95% confidence intervals for the entire simulation period. Intensity was log-transformed to conform to the assumptions of simple linear regression.

RESULTS

Fire frequency

Within each level of fire suppression, mean fire rotation was shorter with higher temperatures (Table 3), reflecting an increase in fire frequency and area burned. For all fire attributes, differences among precipitation levels for a given temperature and suppression level were small compared to differences due to temperature or fire suppression level, and for simplicity, only values for current precipitation levels are referenced in this section. Under 90% fire suppression, warming of 3° and 6° C reduced the fire rotation from 432 years at current temperatures to 282 and 176 years, respectively, representing a 35% and 60% decrease compared to current climate. With 10% fire suppression, 3° and 6° C of warming reduced the fire rotation from 57 years to 31 and 21 years, respectively, representing a 46% and 63% decrease in fire rotation.

The mean number of fire years was higher with warming, and the relative increases were especially large with sustained 90% fire suppression. With current temperatures and 90% fire suppression, there were an average of 56 fire years in the 500-year simulation; with +3° and +6° C of warming, there were an average of 80 and 104 fire years, respectively, representing an increase of 43% and 86%. With only 10% of fires suppressed, there were more fires overall and more fire years gained for each step in warming, but the relative change due to warming was comparatively less dramatic: with current temperatures, there were 302 fire years out of the 500 simulation years, which increased to 355 and 391 fire years under +3° and +6° scenarios, respectively. This represented increases of 18% and 28%.

Fire intensity

Area-weighted mean fire intensity was not strongly affected by changes in precipitation, and remained relatively high over time under scenarios with sustained 90% fire suppression and no change in temperature, or only 3° of warming (Figure 3). With sustained 90% fire suppression and 6° of warming, mean fire intensity decreased over time, falling off sharply for the first 200 years. With only 10% of fires suppressed, mean fire intensity was lower for a given temperature than under 90% fire suppression, and in all temperature scenarios with 10% fire suppression, mean intensity decreased until year 200 and then stabilized.

Averaged over the entire simulation time, mean fire intensity in +0° and +3° scenarios with 90% fire suppression were essentially equal, while mean fire intensity in the +6° scenarios was ~32% lower than in +0° and +3° scenarios (Table 3). In 10% fire suppression scenarios, fire intensity averaged over the entire simulation decreased with each increase in temperature, such that fire intensity in the +3° and +6° scenarios averaged 18% and 39% lower than the current climate scenario, respectively.

Fuel loading

Patterns in fire intensity over time were roughly mirrored by a reduction in fuel loads that accompanied warming and low fire suppression (Figure 4). With 90% of fires suppressed, mean fuel loads remained high under current temperatures and 3° of warming, but decreased somewhat over time under 6° of warming. Under 10% fire suppression, fuel loads were considerably lighter than under 90% suppression for a given temperature scenario, lower under higher temperatures, and remained relatively stable over time.

Fire severity

As with fire frequency and intensity, effects of changes in precipitation on severity trends were small relative to effects of temperature and fire suppression. Unlike mean fire intensities, mean percentage of area that burned with more than 70% tree mortality (high severity) tended to increase under higher temperatures and low fire suppression (Figure 5).

In all temperature scenarios with 90% fire suppression, approximately 10% of area that burned during the first 100 years burned at high severity, and with no temperature increase, the percentage of area burned at high severity remained relatively unchanged for the remainder of the simulation. With 3° of warming, percentage of area burned at high severity increased only slightly over time. With 6° of warming, it increased sharply through year 300, stabilizing at approximately 35-40% of burned area over the remainder of the simulation period. Averaged over the entire simulation, 3° and 6° of warming increased the mean percentage of fire area classified as high severity by approximately 7% and 19%, respectively, compared to current climate (Table 3).

In scenarios with only 10% of fire suppressed, the percentage high-severity fire area under current temperatures or 3° of warming was similar to or slightly greater than values for those climate scenarios under 90% suppression, and tended to increase slightly over time. Over the first 100 years, the percentage of area burned at high severity for both of these climate scenarios was approximately 15%, progressing over time to ~10-20% and 23-30% for current and +3° temperature scenarios, respectively. For the +6° scenarios, the percentage of burned area classified as high severity was higher (~20%) over the first 100 years than for other scenarios, indicating that these scenarios had more high-severity burned area early in the simulation. This percentage increased over time, though less steeply than the same climate scenarios under 90% suppression. Over the final 100 years, approximately 35% of burned area was classified as high severity, which was similar or slightly less than for the same time period and temperature scenarios under 90% fire suppression. Over the entire simulation, +3° and +6°

scenarios had a mean of ~9% and 13% more area classified as high severity, respectively, than the current climate scenario.

Vegetation

Contrasting relationships among fire intensity, fuel loadings, and fire severity suggest that fire interactions with vegetation on the landscape changed over time (Figure 6). Under 90% fire suppression and current temperatures, non-forest (area with no basal area or trees per ha, which may be grassland or shrubland) and area dominated by small trees (<12.7cm diameter at breast height) each comprised less than 5% of the landscape at all time steps. With 3° of warming, the prominence of non-forest was unchanged, while the percentage of the landscape dominated by small trees was variable over time but generally higher than for current temperatures, reaching approximately 10% of the landscape by year 500. With 6° of warming, non-forest remained an insignificant fraction of the landscape, but area dominated by small trees increased, rising over time from 30% to ~38% of the landscape.

The distribution of these vegetation types under current temperatures was relatively similar under 10% and 90% fire suppression. In +3° scenarios, non-forest increased slightly over time but remained less than 5% of the landscape, while the prominence of small trees was somewhat higher than with 90% suppression, ranging from ~10-15% of the landscape over time. With 6° of warming, small trees occupied 35-40% of the landscape, which was similar to their response with 90% fire suppression; however, with 10% fire suppression the percentage of the landscape they occupied tended to decrease slightly over time, rather than increasing. Non-forest expanded steadily with 6° of warming and 10% fire suppression, growing from less than 5% of the landscape at year 100 to 15% of the landscape by year 500.

Discussion

The relationships illustrated here reflect complications inherent in estimating future fire dynamics on this landscape. By incorporating both empirical and mechanistic strategies to estimate the response of fire and vegetation dynamics at high-resolution landscape scales, simulation models like FireBGCv2 provide a valuable complement to short-term field studies and studies that are unable to investigate multiple attributes of potential future ecological conditions (Keane et al. 2004, He et al. 2008).

Fire rotations on this simulated landscape decreased under scenarios of climate warming, with concurrent increases in cumulative area burned and number of fire years. Even with 90% of fires suppressed, the fire rotation with 6° of warming was 296 years shorter than with current temperatures, which vividly illustrates the degree to which changes in climate may impact fire regimes on this landscape. These results agree with widely held expectations that warming temperatures will increase fire frequency in the West by lengthening fire seasons and reducing fuel moistures, increasing the frequency of ignitions (McKenzie et al. 2004, Westerling et al. 2006). Such climate-mediated changes in fire regimes will act in concert with direct effects of warming on plant physiology to impact fire behavior and vegetation dynamics (Lenihan et al. 2003, Littell et al. 2010). Unfortunately it is impossible to entirely isolate the effects of climate and fire in these simulations, since even scenarios with high fire suppression experienced higher fire frequency with climate warming, but comparing scenarios outcomes is still instructive.

These simulations also emphasize the importance of temperature and fire relative to precipitation on this landscape. Results for different precipitation scenarios sometimes hinted at a trend, but differences were generally too small and variable to be conclusive. This may appear somewhat surprising, since precipitation contributes to both vegetation and fire dynamics by affecting productivity and fuel moistures. However, fire season on this landscape (summer and early fall, nearly the same as the growing season) has always been dry, and is likely to remain so. Changes in precipitation are expected (and simulated here) to be largest for fall, winter, and spring seasons (Mote and Salathé 2010), leaving precipitation during the summer season, which is already negligible, relatively unchanged. Consequently, changes in growing season water available to vegetation may mainly reflect changes in temperature, as warmer summers dry and heat the landscape simultaneously, confounding the effects. Similarly, climate change will probably affect fuel moisture mainly by way of temperature, as warmer temperatures dry the landscape earlier and keep it dry longer each year (Westerling et al. 2006). Also, these simulations do not include increased weather variability that is expected to accompany climate change (Mote and Salathé 2010). Fluctuations such as lengthened wet/ dry cycles may have more potential to alter fire regimes than is represented here (Holden et al. 2007).

Fire intensity and fuels

In all scenarios, high mean fire intensities early in simulation time reflected consumption of current heavy fuel loads. Under sustained high fire suppression, high fire intensities were maintained under current temperatures and 3° of warming, as fuel loads were allowed to persist. However, with few fires suppressed, fire intensities decreased early in the simulation, along with fuel loads. Allowing this early "burn off" resulted in mean fire intensities that were approximately 28% and 41% lower for current temperatures and 3° of warming, respectively, over the entire simulation. Response of mean flame lengths closely tracked trends in fire intensity (Appendix Figure A1). Because wind speeds in FireBGCv2 are only permitted to vary within 50% of values set at the site level, these simulations lack extreme fire weather conditions that lead to high fire intensities and long flame lengths characteristic of extreme fire events in this ecological system; instead, differences in fire behavior among scenarios mainly reflect differences in fuel loads.

Simulated decreases in future fire intensities contradict some previous work that suggests warming is likely to promote high-intensity fires (Flannigan et al. 2009, Hessl 2011). This idea is based on the assumption that warming will increase net primary production, thereby increasing the rate at which fuels accumulate (e.g., Lenihan et al. 2008a, Rogers et al. 2011). Although potential maximum net primary production should increase under higher temperatures, the degree to which productivity actually increases will likely depend on precipitation constraints (Lenihan et al. 2008a), especially in areas that are already water stressed during summer droughts. In these simulations, net primary production was essentially the same in +0° and +3° scenarios, and considerably lower in +6° scenarios (Appendix Table A4), indicating that even a 10% increase in precipitation was evidently insufficient to overcome moisture deficits brought on by warming on this landscape. Any productivity gains realized in higher, colder forests appear to have been outweighed by losses in low-elevation dry forests. Along with increased fire frequency, decreasing forest productivity reduced fuel loads, and consequently reduced mean fire intensities.

Fire severity—interactions with vegetation

Under current climate and high fire suppression, simulated percentage of area burned at high severity should roughly reflect current real-world fire trends, but was lower than has been measured over several recent fires near the study area (~8% vs. ~21% averaged over forest types; Meigs et al. 2009). This probably reflects differences in classification strategies as well as patterns of real-world fire suppression that are not duplicated in the model. Fire suppression in reality is non-random. Fires that ignite during mild weather are often quickly extinguished, while fires that occur during extreme weather—which are prone to burn at high intensity and cause extensive tree mortality—are more difficult to contain and extinguish. In FireBGCv2, however, fire suppression is random, and does not depend on fire intensity. The percentage of high-severity burned area is therefore low because the total burned area includes relatively more low-severity fires than would be permitted to burn in reality. Consequently, it is possible that these simulations underestimate the relative prominence of high-severity fire.

At first glance, it may appear counterintuitive that when averaged over time, less fire suppression corresponded to relatively more area burned at high severity, as was observed in the +0° and +3° scenarios. Conceivably, as fuel loads diminish with more frequent future fires, fires could become less intense, and in those less-intense fires the percentage of fire area with high tree mortality could decrease (Hessl 2011). However, for a given climate scenario over time, lower mean percentage of high-severity fire area did not correspond to lower mean fire intensities or to lighter fuel loads. Instead, fire severity reflected interactions with changing vegetation composition and structure on the landscape.

With little fire suppression, high-intensity fires early in these simulations burned off heavy fuel loads, but simultaneously created more area dominated by non-forest or small regenerating trees. Small trees in regenerating stands are more likely to be scorched and killed in subsequent fires, resulting in more burned area that was classified as high severity due to tree mortality. This effect has been noted before in northern California (Miller et al. 2012), where from 1987 to 2008 the percentage of high-severity fires was found to be higher in areas dominated by small-diameter trees. In this way, more fire on the landscape creates a feedback that, while lowering mean fire intensity, raises the proportion of area with trees that are easily killed by fire. While this does not fit the common image of high fire severity as describing extensive death of large, mature trees, it nonetheless suggests that in the future, larger areas of early seral forest structure may lead to increasingly sensitive interactions between vegetation and fire.

Under 6° of warming, there was no significant difference between 90% and 10% fire suppression scenarios in the overall percentage of fire area that burned at high severity. Only the temporal pattern was different: with 90% of fires suppressed, the percentage of area burned at high severity was lower early in the simulation and ended slightly higher than under 10% fire suppression. This suggests a complicated set of interactions on the landscape. Even with 90% fire suppression, fires were considerably more frequent with 6° of warming than under current temperatures. This decreased fuel loads over time, but not to the same extent as with 10% fire suppression. Consequently, fire intensities, though decreasing over time, remained comparatively higher with 90% fire suppression, tending to elevate fire severity. In scenarios with 6° of warming and only 10% of fires suppressed, however, the landscape began to shift to non-forest, and area occupied by small, vulnerable trees began to contract. In Yellowstone National Forest, similar interactions between fire and vegetation have been projected to lead to vegetation type conversion from forest to shrubland within this century (Westerling et al. 2011). Likewise, although dry forest types like ponderosa pine are adapted to fire, work by Savage and Mast (2005) suggests that repeated burning may cause conversion of these forests to shrubland, increasing the prominence of non-forest vegetation on the landscape. With less area occupied by small trees vulnerable to fire mortality, the percentage of fire area burning at high severity under high temperatures and very frequent fire may have been dampened as simulations progressed.

The potential for dramatic vegetation type conversions from forest to grass- or shrubland may seem ominous, but they are not necessarily negative ecological developments for this landscape. Baker (2012) argues that high-severity fire was historically more prevalent in dry forests than is generally believed, which could mean that widely held concerns about modern stand-replacing fires in dry forests are unfounded. Simulations of historical variability in the Deschutes National Forest found that prior to Euroamerican settlement, approximately 2-18% of the landscape was grass- or shrubland (Agee 2003, Kennedy and Wimberly 2009), 25% of the landscape was occupied by early successional forest, and 20-30% of fires were stand replacing (Kennedy and Wimberly 2009). Such conditions are not dissimilar from patterns simulated here under potential future climate and fire suppression scenarios, suggesting that a climate-spurred resurgence of fire may actually return the fire regime on this landscape to a more historical range of variability.

Conclusions and implications

Extensive research agrees that the frequency of fire and total area burned are likely to increase in response to climate warming, and my simulations support those conclusions. However, less attention has been paid to the potential character of future fires, a deficiency that this study attempts to remedy. The tendency for fire to become more frequent with warming may worry land managers, but the accompanying decrease in fire intensity in these simulations is illuminating. It suggests that—unless large increases in precipitation permit rising forest productivity—the more fire a dry landscape like this one experiences, the less catastrophic those fires will be, even with considerable climate warming. By allowing fires to burn whenever possible (wildland fire use), or employing fire surrogates to reduce fuels, managers may be able to modulate the intensity of future fires so that they can be more readily controlled when needed. In these scenarios, sustained fire suppression only maintained an unstable state on the landscape for a longer period, exacerbating fire intensity and slowing or precluding the establishment of a less intense fire regime.

Unfortunately, the early portions of these simulations suggest that the intense fires recently experienced near the study area may remain common until landscape fuels and vegetation come into balance with the fire regime, and attain a more stable condition. On the other hand, decreases in net primary production associated with warming may have the unexpected beneficial effect of helping to limit fuel accumulation, rendering future fires less intense. Interactions between fire, climate, and vegetation in these results also suggest that with warming, the impact of fire on the landscape will become increasingly interrelated with vegetation structure, as more fire promotes and is promoted by grass- and shrubland and early seral forest structure (Hessburg et al. 2005). These new realities would require adjustments on the part of land managers: more-frequent fire and wider distributions of young vegetation structures may not fall outside the landscape's historical range of variability, but they could be undesirable from social or economic standpoints, and management goals may need to be amended to account for them.

LIMITATIONS

In most modeling studies, including this one, the value of the simulations lies less in the absolute numbers they produce than in comparison among potential scenarios, and the information those comparisons provide regarding potential ecosystem development and function. Like all modeling projects, this study is limited by the data used for parameterization and by the assumptions of the modeling platform.

For example, this parameterization of FireBGCv2 omits insects and pathogens, such as mountain pine beetle (*Dendroctonus ponderosae*), which have wide-ranging effects on East Cascades landscapes and may interact unpredictably with changes in climate and fire regimes (McKenzie et al. 2004, Littell et al. 2010, Hicke et al. 2012); potential management actions like logging, thinning, and prescribed burning are also excluded.

Each model also has its own quirks; for example, although fire behavior in FireBGCv2 responds to available fuel and fuel conditions, simulated fire spread relies only on vectors of wind and slope. Consequently, it was difficult to model historical fire regimes on this landscape, where extreme temperature and fuel moisture gradients can exert considerable control over fire spread. Fuel moisture conditions are simulated at a coarse (site) level, so stand-level fuel moisture variability arising from differences in aspect and slope are largely unaccounted for.

Tree growth algorithms in FireBGCv2 are species specific, but four minor species in this study (sugar pine, incense-cedar, noble fir, and giant chinquapin) were not present in the model due to a lack of parameters and algorithms, so algorithms for similar species were used as surrogates. Also, species presence in the model is limited to those species included at the beginning of the simulation; the model cannot simulate the immigration of species from adjacent systems. In particular, it is possible that higher temperatures on this landscape would lead to juniper encroachment from the east, especially with continued fire suppression. Finally, shrub species are not individually parameterized in this implementation of FireBGCv2. Each shrub species interacts differently with fire according to its physiology and life history (e.g. flammability, sprouting ability), and characteristics of shrub communities can control local fire regimes (Keeley et al. 2008), but these dynamics are not captured here.

More generally, the use of patches to represent biophysical settings and community types in maps and models—as in this study—is widespread, but patches and patch-based metrics are increasingly viewed as inadequate representations of ecosystem attributes that naturally occur as gradients (Cushman et al. 2008b, McGarigal et al. 2009).

FIGURES AND TABLES

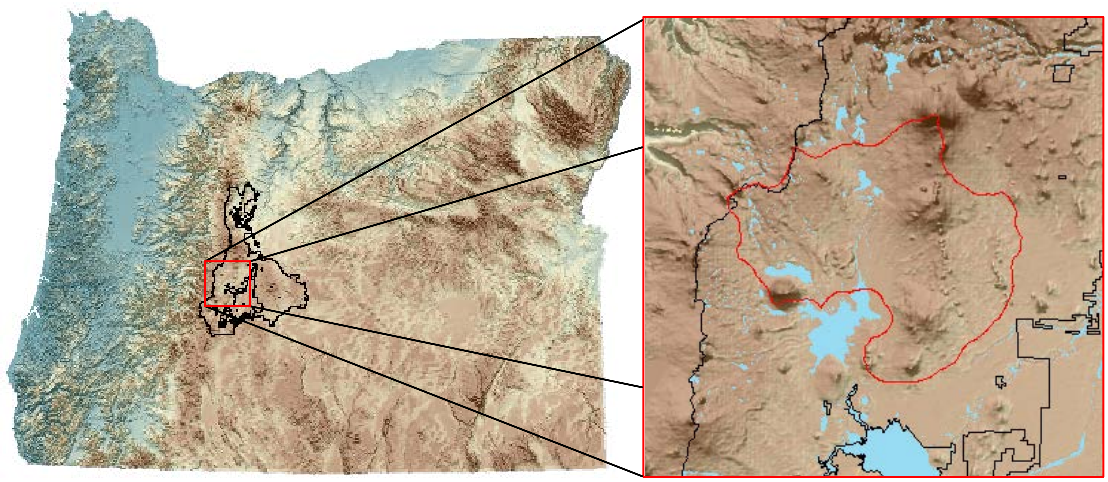

Figure 1. Study area in Deschutes National Forest, Oregon.

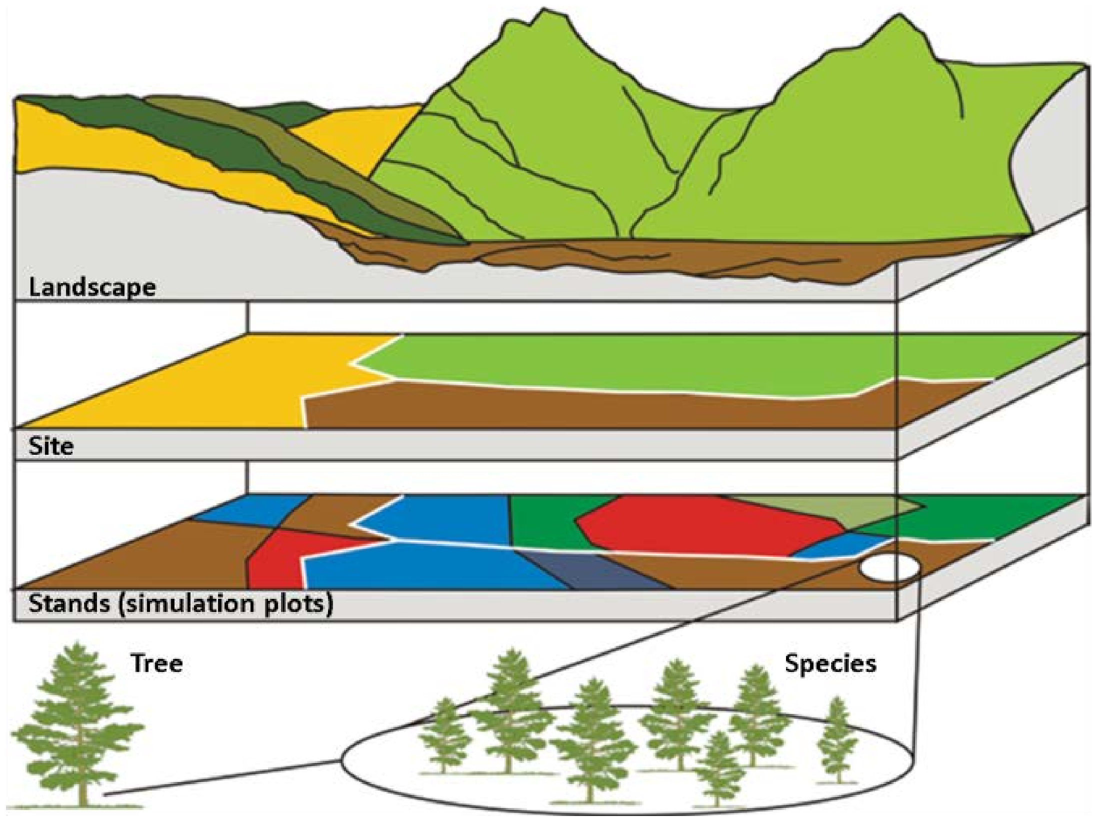

Figure 2. The five organizational scales of FireBGCv2. From Keane et al. (2011), used with permission.

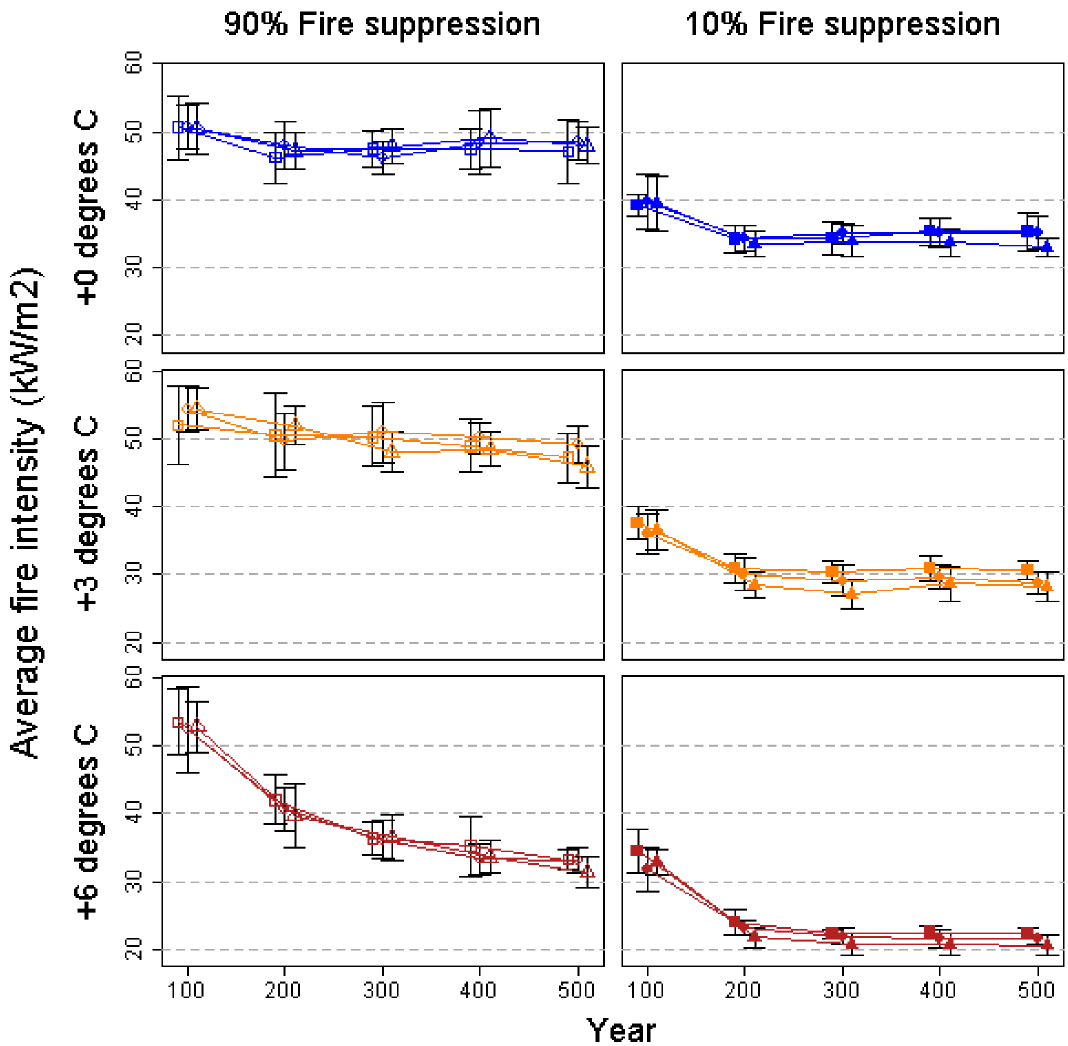

Figure 3. Area-weighted mean fire intensity for each scenario over time. Whiskers are standard deviation. Values represent averages over the prior 100 years. Circles: current precipitation; squares: +10% precipitation; triangles: -10% precipitation.

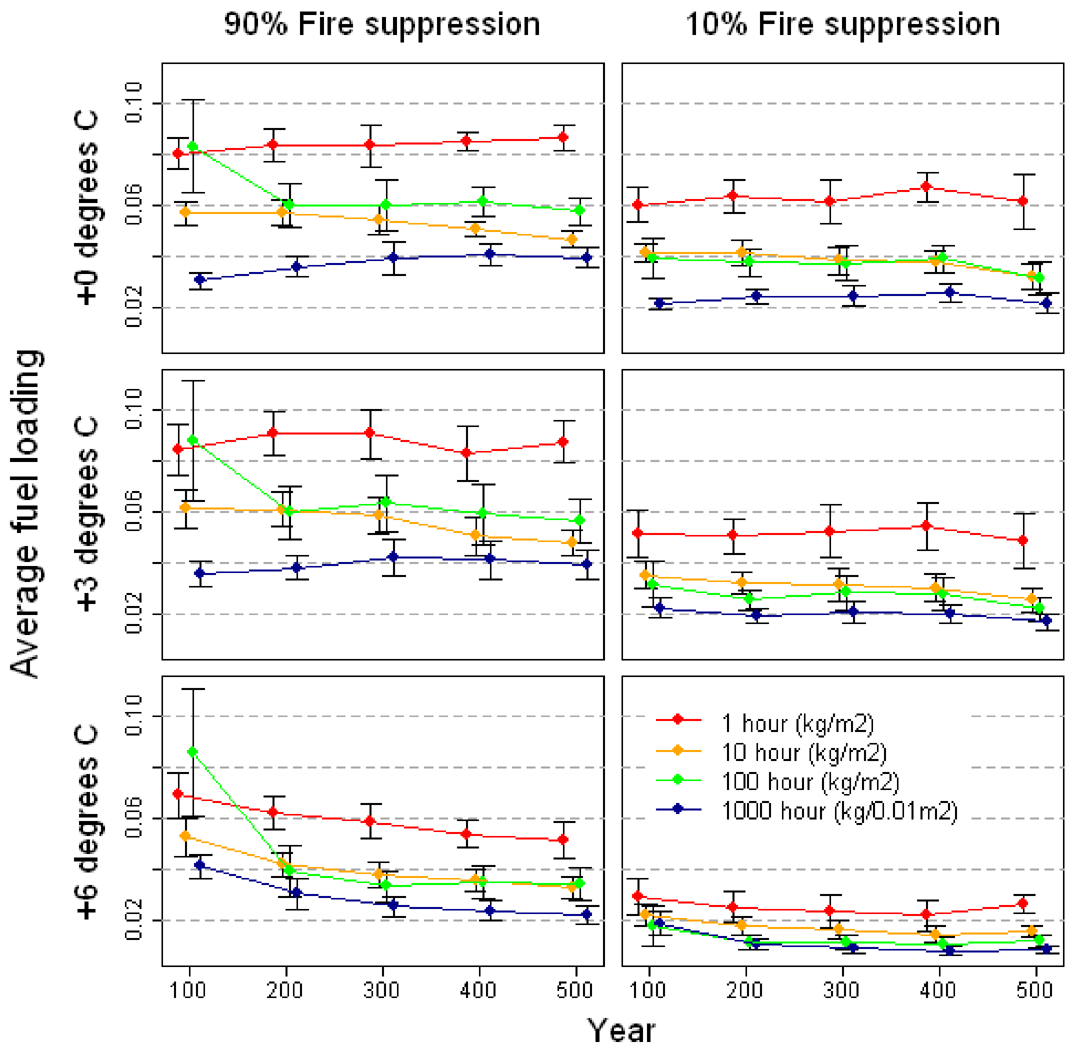

Figure 4. Area-weighted mean fuel loadings over time. Whiskers are standard deviation. Precipitation levels were not notably different from one another and were pooled for display. 1000-hour fuels are scaled differently from other fuels to simplify display.

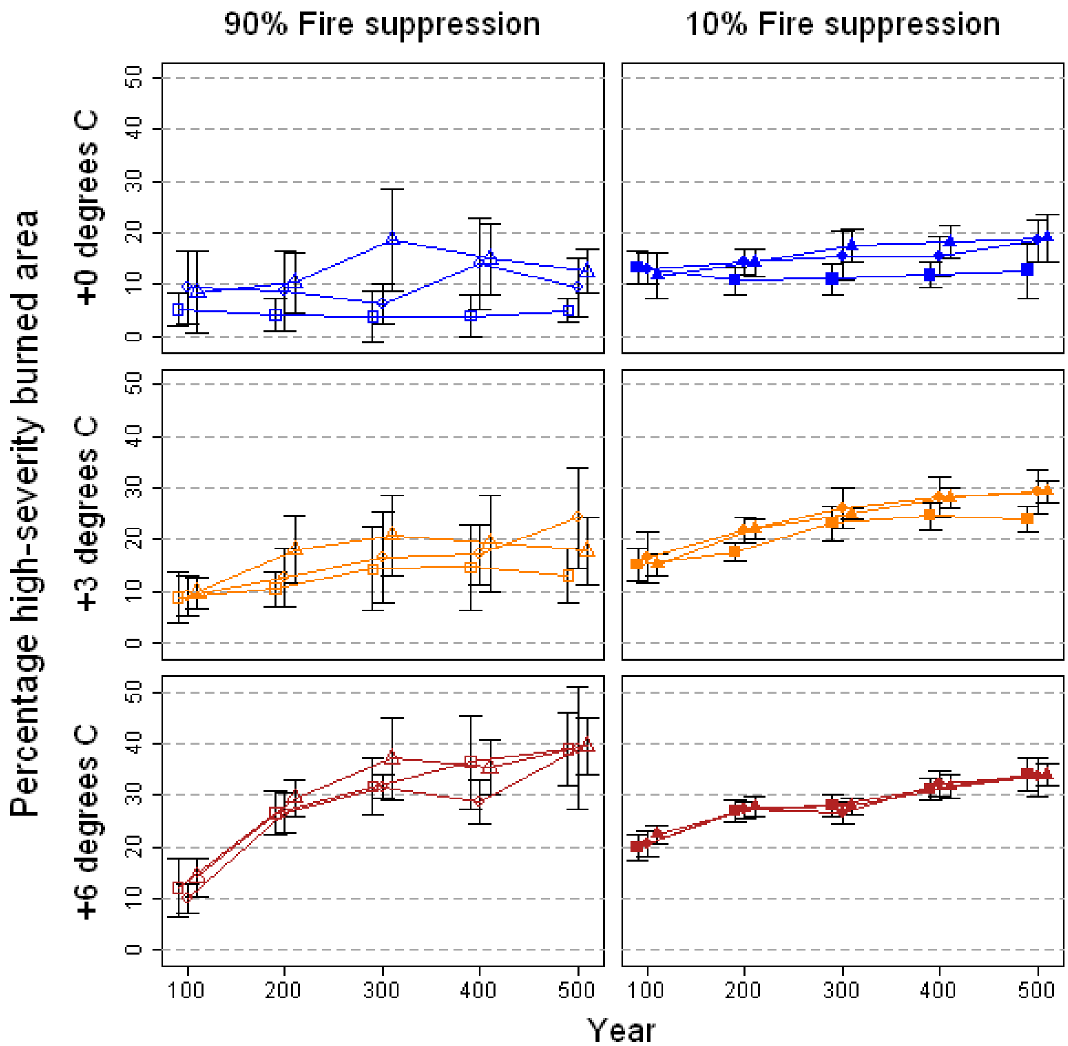

Figure 5. Mean percentage high-severity burned area for each scenario over time. Whiskers are standard deviation. Values represent averages over the prior 100 years. Circles: current precipitation; squares: +10% precipitation; triangles: -10% precipitation.

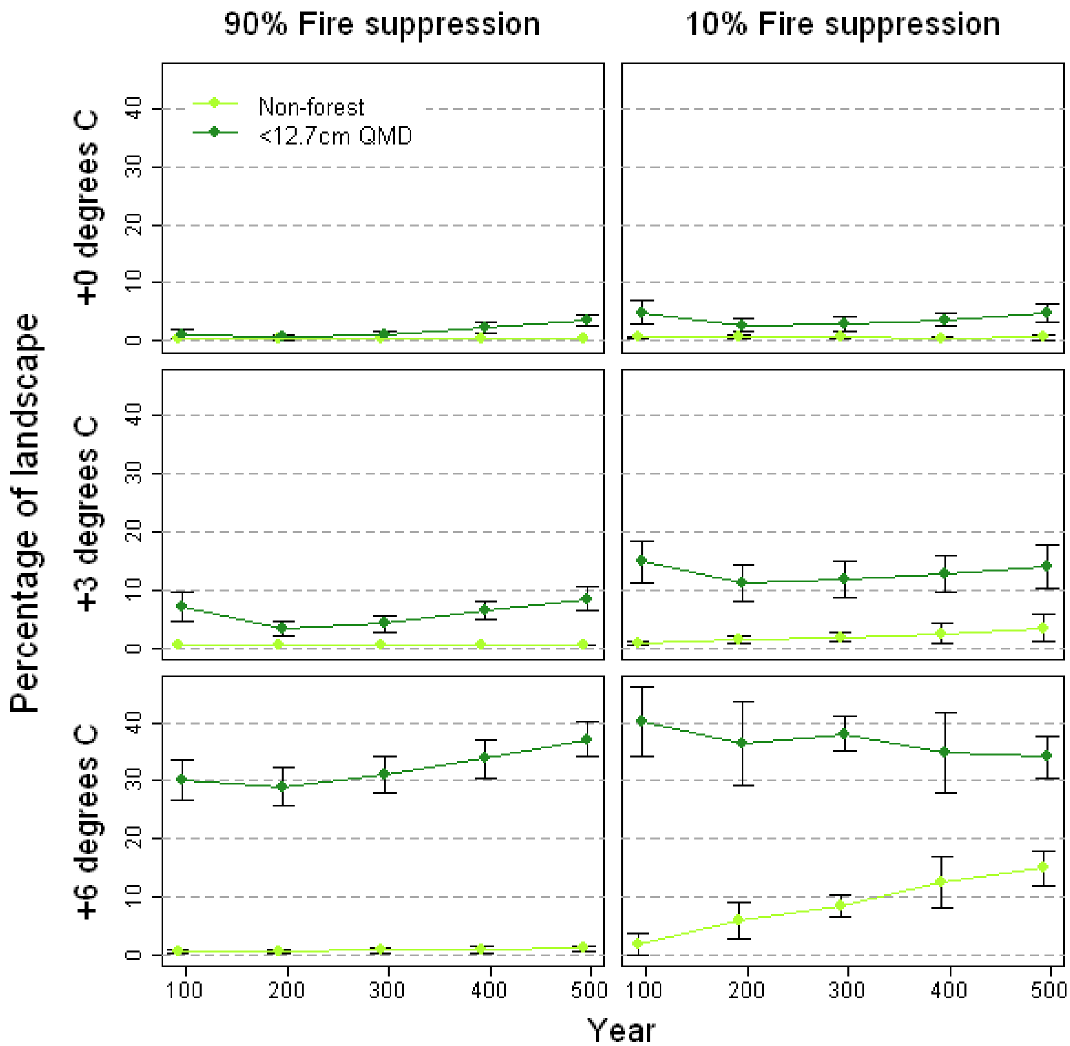

Figure 6. Mean percentage of the landscape dominated by small trees (<12.7 cm QMD) and non-forest. Whiskers are standard deviation. Precipitation levels were combined for a given temperature and fire combination.

Table 1. Process simulation scales in FireBGCv2. Adapted from Keane et al. (2011).

Organizational scale	Description	Processes simulated
Landscape	Extent of simulation area	Fire ignition and spread, seed dispersal
Site	Homogeneous biophysical settings	Weather, soils, fuel moistures
Stands (plot)	Vegetation communities	Photosynthesis, respiration, evapotranspiration, decomposition, fire behavior and effects
Species	Tree, shrub, and grass species	Phenology, regeneration, carbon allocation
Tree	Individual tree characteristics	Mortality, growth, litterfall

Table 2. Simulated climate scenarios. Values for temperature represent final offsets from the historical weather stream in degrees Celsius; values for atmospheric CO_2 represent the final atmospheric CO_2 concentration. Values for precipitation represent multipliers applied to the historical weather stream.

	Final annual offset	Final seasonal offset			
Temperature/CO_2 (° Celsius/ppm)		Winter	Spring	Summer	Fall
Current	0°/390ppm	0°	0°	0°	0°
Warm	3°/550ppm	2.866°	2.687°	3.582°	2.866°
Hot	6°/800ppm	5.731°	5.373°	7.164°	5.731°
Precipitation (Scalars)					
Dry	0.90	0.948	0.948	0.768	0.937
No change	0	1	1	1	1
Wet	1.10	1.158	1.158	0.938	1.145

Table 3. Mean number of fire years, cumulative area burned, fire rotation, fire intensity, and percentage of fire area burned at high severity over 500-year simulation period. Letters indicate overlapping 95% confidence intervals for mean intensity and percent high-severity fire area.

		# Fire years*	Cumulative area burned (ha)*	Fire rotation (years)*	Average fire intensity (kW/m^2)**	% Fire area burned at high severity**
90% Fire suppression						
	Dry	57 (9)	52,193 (18,181)	449 (165)	47.9[a] (46.1, 49.9)	12.3[o,r] (11.0, 13.7)
+0° C	*No change*	56 (5)	61,908 (31,544)	432 (246)	48.6[a] (46.7, 50.5)	8.1[n] (6.8, 9.4)
	Wet	55 (5)	53,651 (18,344)	423 (121)	46.9[a] (45.0, 48.8)	3.8[m] (2.5, 5.2)
	Dry	83 (7)	95,321 (21,815)	228 (48)	45.8[a] (44.0, 48.6)	16.9[p,q] (15.6, 18.3)
+3° C	*No change*	80 (9)	84,796 (29,208)	282 (127)	49.0[a] (47.1, 51.0)	15.5[p,q,r] (14.2, 16.8)
	Wet	78 (7)	101,662 (36,927)	227 (73)	47.1[a] (45.3, 49.0)	12.8[o,p,r] (11.49, 14.2)
	Dry	112 (10)	147,886 (26,992)	145 (24)	31.3[c,f,g] (30.1, 32.6)	30.4[u,w] (29.1, 31.7)
+6° C	*No change*	104 (9)	128,841 (38,896)	176 (53)	33.1[c,f,g] (31.8, 34.4)	27.2[u,v] (25.9, 28.5)
	Wet	107 (8)	153,650 (33,090)	141 (30)	33.2[f,g] (32.0, 34.6)	28.6[u,v,w] (27.2, 29.9)

* Standard deviation in parentheses.

** 95% Confidence interval in parentheses.

Table 3 (Continued).

		# Fire years*	Cumulative area burned (ha)*	Fire rotation (years)*	Average fire intensity (kW/m^2)**	% Fire area burned at high severity**
10% Fire suppression						
	Dry	306 (6)	413,126 (23,995)	51 (3)	32.9[b,c,f,g] (31.7, 34.3)	16.2[p,q] (14.9, 17.6)
+0° C	*No change*	302 (8)	373,757 (58,853)	57 (9)	35.0[b,f] (33.6, 36.4)	15.0[p,q,r] (13.7, 16.4)
	Wet	294 (12)	404,672 (50,456)	52 (6)	35.2[b,f] (33.8, 36.7	11.8[o,r] (10.5, 13.2)
	Dry	363 (9)	723,586 (84,032)	29 (3)	28.1[d,e] (27.0, 29.3)	24.0[t] (22.6, 25.3)
+3° C	*No change*	355 (13)	689,150 (87,267)	31 (4)	28.7[c,d,e] (27.6, 29.9)	24.3[t] (23.0, 25.6)
	Wet	359 (8)	661,126 (76,216)	32 (4)	30.7[c,d,f,g] (29.5, 31.9)	20.8[s] (19.5, 22.2)
	Dry	399 (12)	1,026,373 (97,784)	20 (2)	20.6[i,j] (19.8, 21.4)	28.3[u,v,w] (27.0, 29.6)
+6° C	*No change*	391 (8)	987,419 (78,145)	21 (2)	21.5[h,I,j] (20.7, 22.4)	27.6[u,v] (26.3, 28.9)
	Wet	394 (10)	990,508 (71,489)	21 (2)	22.4[h,i] (21.6, 23.4)	27.7[u,v] (26.3, 29.0)

* Standard deviation in parentheses.

** 95% Confidence interval in parentheses.

Chapter 3—Forest Vegetation & Landscape Dynamics

Abstract

Climate and fire are major drivers of vegetation composition and structure in forests of the Oregon East Cascades. Climate change and associated changes in fire regimes are widely expected to affect the form and function of East Cascades landscapes, complicating efforts to estimate future conditions in forests already altered by 20th-century Euroamerican land and fire management. Since climate change will affect forests both directly via physiology and indirectly via disturbance regimes, projecting its ultimate effect on the landscape requires tools that can estimate complex ecological interactions.

I used FireBGCv2, a spatially explicit mechanistic forest succession and disturbance model, to simulate fire, vegetation, and landscape dynamics in a portion of the Deschutes National Forest under 18 potential climate change scenarios. Simulation scenarios included all combinations of +0°C, +3°C, and +6°C of warming; +10%, ±0%, and -10% historical precipitation; and 10% and 90% fire suppression, and were run for 500 years. Ordination with nonmetric multidimensional scaling (NMS) was used to holistically assess potential differences in landscape composition, structure, and configuration among scenarios over time. Distribution of vegetation types and structural stages were mapped for each climate and fire suppression scenario.

Changes in precipitation did not strongly affect vegetation or landscape dynamics. Following shifting locations of suitable environmental conditions, warm forests of mixed conifer and ponderosa pine (*Pinus ponderosa*) migrated upslope under warmer temperatures, regardless of fire suppression, displacing cool wet conifer forests at high elevations. With 3° of warming, cool wet conifer forests were reduced from approximately 37% to 10% of the landscape; with 6° of warming, they were essentially eliminated. Warmer temperatures and less fire suppression promoted the expansion of non-forest (<10% forest cover) on the landscape, such that under 3° of warming non-forest increased from 2-3% of the landscape to 5% and 13% of the landscape with 90% and 10% fire suppression, respectively, and 18% and 38% of the landscape under 6° of warming and 90% and 10% fire suppression, respectively. Because warm conifer forests were able to move upslope, their relative coverage on the landscape was less affected than cool conifer forests.

Warming and additional fire generally improved the diversity of forest structure, moving the landscape closer to a historical range of conditions by increasing coverage by early successional forests, especially in lower elevations. Scenarios with less fire suppression led to a more heterogeneous, divided landscape than scenarios with sustained high fire suppression. Three degrees of warming did not affect the percentage of the landscape occupied by the oldest, largest forest structure, but under 6° of warming, expansion of non-forest reduced old forest structure by 10-20% compared to current temperatures.

The NMS ordination highlighted the effect of fire on the landscape and the considerable overall difference between the +6° scenarios and the other two scenarios. Differences in composition, forest structure, and landscape pattern that arose under extreme warming suggest the presence of an ecological threshold between 3° and 6° of warming. The magnitude of simulated changes suggests that future land and fire management will need to be responsive and adaptive in order to plan and implement realistic goals on this landscape.

INTRODUCTION

Ecosystem composition and function arise from interactions among climatic, biological, and anthropogenic forces. On the dry east side of Oregon's Cascade Mountains, Euroamerican land management over the past century has resulted in significant and ongoing alterations to historical forest structure and processes (Hessburg and Agee 2003, Hessburg et al. 2005, Westerling et al. 2006), but the complex nature of ecological interactions limits our understanding of the ways in which these forests may develop in the future. Potentially significant changes in regional climate are expected within this century (Mote and Salathé 2010), further complicating efforts to characterize potential forest trajectories. Climate change should play a particularly important role in determining future forest dynamics, since climate and its variability are fundamental drivers of forest development and disturbance events that shape the landscape (Westerling et al. 2003, Littell et al. 2009).

Twentieth-century changes

Researchers largely agree that the present state of dry forests in the East Cascades differs dramatically from their condition prior to Euroamerican settlement, when large portions of mid- and low-elevation forests were described as a mosaic of forest types and successional stages (Agee 2003, Hessburg and Agee 2003, Hessburg et al. 2005, Spies et al. 2006). Many stands were open and "park-like" with large, widely spaced ponderosa pines (*Pinus ponderosa*) and sparse understories maintained by frequent, low-intensity surface fires, a structural condition that is now rare (Everett et al. 2000, Hessburg and Agee 2003, Hessburg et al. 2005, Kennedy and Wimberly 2009, but see Baker 2012). Area dominated by the smallest structural classes has also declined (Hessburg et al. 2005). A century of fire exclusion, grazing, and logging in these forests has led to a narrowed age-class distribution, surface and ladder fuel accumulation, and the loss of open, park-like stands, as fire-sensitive tree species like grand fir (*Abies grandis*) replace fire-tolerant species and fill in across an increasingly homogeneous landscape (Hessburg and Agee 2003, Perry et al. 2004, Hessburg et al. 2005, Spies et al. 2006).

This loss of landscape heterogeneity has destabilized disturbance regimes. Fire previously maintained a heterogeneous, self-perpetuating mosaic of diverse cover types and age classes by creating natural fire breaks and modulating the spread of other contagious disturbance events such as insect outbreaks (Everett et al. 2000, Hessburg et al. 2005, Spies et al. 2006). By simplifying forest structure, altering understory composition, and severely restricting fire occurrence, Euroamerican land management has promoted unusually large areas of stressed, vulnerable forest (Hessburg et al. 2005, Moeur et al. 2005, Naficy et al. 2010). These changes have created a tension between forest structure and fire regime: in their dense, homogeneous, fire-sensitive condition, East Cascades forests now resemble forests like those of western Oregon, which more commonly experience low-frequency, high-severity fire events; however, the arid interior setting still promotes frequent, climatically driven wildfires (Hessburg et al. 2005, Littell et al. 2009, Haugo et al. 2010). As a result, wildfires, when they do occur, now tend to be high-severity, stand-replacing events—a deviation from historical landscape processes that has serious implications for wildlife habitat and other forest resources and services (USFS 2004, Moeur et al. 2005, USFS 2005, Spies et al. 2006, Kennedy and Wimberly 2009).

Climate change and fire suppression effects

Climate change is likely to complicate efforts to understand the trajectories of these altered forests. Within the next century, the Pacific Northwest is projected to experience warmer temperatures in all seasons, with estimates for annual mean temperature increases ranging from approximately 4.5-7° C by the end of the 21st Century (Mote and Salathé 2010). Estimates of changes in precipitation are more variable and less certain, but small increases in winter and spring precipitation and decreases in summer precipitation are expected in the interior Northwest (Mote and Salathé 2010). With warming temperatures, more winter precipitation will fall as rain, resulting in earlier snowmelt and significant decreases in snowpack, dry-season runoff, and growing-season soil moisture (Leung et al. 2004, Elsner et al. 2010).

Climate affects vegetation directly, via species physiology and phenology (Rehfeldt et al. 2006, Littell et al. 2010, Chmura et al. 2011), and indirectly, by mediating disturbance regimes (McKenzie et al. 2004, Westerling et al. 2006, Haugo et al. 2010, Chmura et al. 2011). Whether the direct (i.e. physiological) or indirect (i.e. disturbance) effects of climate change play a larger role in shaping future forest characteristics may depend on site-specific attributes such as elevation and related temperature and moisture gradients, and interactions with species tolerances and disturbance regimes (Littell et al. 2009, Haugo et al. 2010).

Tree physiology and phenology will be affected by climate change mainly via increasing temperatures and carbon dioxide (CO_2) fertilization, which have the potential to increase forest productivity in the West (Bachelet et al. 2001, Latta et al. 2010). Phenology will likely respond in species-specific, non-linear ways; for example, moderate warming may advance bud break in Douglas-fir (*Pseudotsuga menziesii*), but too much warming may lead to delayed bud burst due to insufficient chilling (Chmura et al. 2011). CO_2 fertilization should improve photosynthetic and water use efficiency, but it is unclear whether acclimation will eliminate these benefits with time (Chmura et al. 2011). Although warmer temperatures may initially increase forest productivity in high-elevation forests currently limited by short growing seasons (Latta et al. 2010), severe warming may increase moisture deficits and drought stress, especially in moisture-limited lower-elevation forests, reducing productivity there (Rehfeldt et al. 2006, Littell et al. 2010). Vegetation in some areas is already stressed by changes in local conditions that are likely to be permanent (Waring et al. 2011).

The cumulative effect of these interactive factors on individual species and communities is difficult to project (Williams et al. 2007), but is likely to result in shifts in the geographic and elevational ranges of tree species and related forest types (Rehfeldt et al. 2006, Coops et al. 2010, Littell et al. 2010, Coops and Waring 2011). Forests of ponderosa pine (*Pinus ponderosa*), a major ecological and economic species on the East Cascades landscape, are currently hemmed into a narrow geographic band by arid juniper woodlands and grasslands to the east and warm mixed conifer forests to the west. If increases in precipitation decrease drought stress in the future, this pine could expand eastward, increasing the overall forested area in the region (Bachelet et al. 2001, Coops et al. 2005). Conversely, if precipitation decreases, or if rising temperatures exacerbate drought stress despite increases in precipitation, ponderosa pine forests will likely contract in the east and expand upslope in the west, with warm mixed conifer advancing ahead of them (Coops et al. 2005, Rehfeldt et al. 2006). Consequently, cold-tolerant forest types such as mountain hemlock and silver fir may be replaced by these heat-tolerant forests (Bachelet et al. 2001, Coops et al. 2005, Rehfeldt et al. 2006).

Similarly, lodgepole pine (*Pinus contorta*) currently occupies a unique niche in East Cascades forests, filling in at high densities as an early seral dominant following stand-replacing disturbance, and persisting in extreme environmental pockets that discourage other species, especially flats and hollows where cold-air drainage suppresses warm-adapted conifers like ponderosa pine (Burns and Honkala 1990). Warming is likely to decrease the prevalence of these cold pockets, leading to more competition with warm-adapted species and potentially a severe decline of lodgepole pine; Coops and Waring (2011) estimate that lodgepole distribution may decline to 17% of its current range by 2080.

The degree to which wildfires are suppressed will also continue to affect distributions of vegetation types in the East Cascades. By altering vegetative successional stages, fire provides opportunities for shifts in forest composition, especially toward shade-intolerant species. Less fire suppression, particularly if combined with higher temperatures, would favor pines: ponderosa pine is both heat- and fire-tolerant, while lodgepole pine, though fire-sensitive, is heat-tolerant and reproduces extensively following fire, thanks to vigorous juvenile growth and serotinous cones in some individuals (Burns and Honkala 1990). However, too-frequent fire may reduce forest cover altogether, promoting shrubs and grasses instead (Savage and Mast 2005). The presence or absence of fire will also have a large impact on future landscape configuration, since fire fragments large areas of homogeneous forest types and ages (Jordan et al. 2008, Naficy et al. 2010).

Estimating future trajectories

A historical park-like state is frequently suggested and sometimes implemented as a restoration goal in East Cascades dry forests (Agee 2003, Hessburg et al. 2005), but future management efforts must contend both with management legacies and uncertain future climate conditions. The complicated interactions between physiology, phenology, and disturbance, as well as the large spatial and temporal scales involved, make these relationships difficult to visualize, quantify, and synthesize (Hessburg and Agee 2003, Agee and Skinner 2005). Additionally, because many factors affecting forest disturbance and development are locally specific (e.g., topography, weather, fuel load, land-use history), landscapes should be considered individually, rather than assigned blanket prescriptions based on a reference system (Perry et al. 2004, Hessburg et al. 2005, Lee and Irwin 2005).

Spatially explicit computer simulations are a powerful tool with which to address landscape-level questions of potential forest development under uncertain future scenarios (e.g. Keane et al. 1999, Keane et al. 2004, McKenzie et al. 2004, Ager et al. 2007, Wimberly and

Kennedy 2008). By incorporating processes of climate, vegetation, and disturbance, simulation models allow researchers to explore potential outcomes of the complex interactions that determine forest trajectories. In this study, I used a spatially explicit mechanistic forest succession and disturbance model to simulate forest development and landscape dynamics in a portion of the Deschutes National Forest under potential climate change and wildfire scenarios. In particular, I explored how major vegetation types on the landscape responded to changes in climate and wildfire, and sought to holistically compare differences in landscape composition, structure, and configuration arising from different scenarios. Estimates of these changes and their impact on landscapes should provide insight for managers and researchers as they seek to understand and plan for future landscape conditions.

Methods

Description of the FireBGCv2 model and information on model parameterization and calibration, as well as descriptions of the study area and simulation scenarios, can be found in Chapter 2 of this thesis.

Analysis—Vegetation types

Tree- and species-level output from FireBGCv2 were generalized into vegetation types in a two-step fashion. First, for the final year (year 500) of each simulation, each species in each stand on the landscape was assigned a stand-level importance value (IV) based on the following formula:

$$\frac{100 \times stand\ \#\ of\ trees\ of\ focal\ species}{stand\ total\ \#\ of\ trees} + \frac{100 \times stand\ basal\ area\ of\ focal\ species}{stand\ total\ basal\ area}$$

Within each stand, each species was then ranked in descending order of importance value. Considering only the species with the highest and second-highest importance values (Sp1 and Sp2), each stand was assigned a preliminary forest type based on a decision tree (Figure 7). Under this system, tree species that are clearly dominant are assigned sole dominance in a stand, while stand with two species that nearly share dominance are assigned hyphenated types. In stands where Sp1 is not the clear dominant but no other species is close to it in dominance (indicating that there are many species present), Sp1 is assigned with a "-MIX" to indicate as much.

When applied to all replicates of all simulation scenarios, this process produced three hundred preliminary vegetation types (including comparatively redundant types such as mountain hemlock-silver fir and silver fir-mountain hemlock), which were then crosswalked to one of eight consolidated vegetation types (Appendix Table A5), in reference to Simpson (2007) and Franklin and Dyrness (1988). Summary statistics were calculated in R (version 2.14.1; www.R-project.org) describing the percentage of the landscape occupied by each forest type for each scenario. To demonstrate geographic shifts, vegetation types were mapped in ArcGIS (version 10) for each replicate of each scenario, and then "averaged" using the Cell Statistics tool in ArcGIS Spatial Analyst to produce maps showing the vegetation type that occurred most frequently in each pixel among the ten replicates of each scenario. Where types tied for majority representation (less than 10% of area for any given map), the Expand tool was used to generalize neighboring vegetation types for clearer display. A similar process was followed to map distribution of forest structural stages, which were directly output from FireBGCv2. Structural stages indicate the dominant tree class in terms of diameter at breast height (DBH) regardless of canopy cover; therefore each may include area that is otherwise classified as non-forest (<10% tree cover).

Analysis—Landscape dynamics over time

Heterogeneity is a notoriously difficult concept to quantify, since it encompasses multiple characteristics of landscape composition, structure, and configuration that are themselves difficult to isolate meaningfully (Cushman et al. 2008a). Multiple metrics can be measured in an attempt to understand and synthesize relationships, but many landscape metrics are correlated, confounding analysis, and it is also difficult to conceptualize changes in multiple variables that are changing simultaneously in space and time (Cushman and McGarigal 2007). Ordination of landscape metrics in multivariate space is a useful method for addressing these complications, because it relies on correlation among variables, instead of requiring orthogonality, and reduces the dimensionality of complex datasets (McCune et al. 2002). Here, I performed a landscape trajectory analysis (as in Cushman and McGarigal 2007, Thompson et al.

2011) using Nonmetric Multidimensional Scaling (NMS) of 28 compositional, structural, and configurational variables to assess dissimilarity among scenarios at 100-year intervals.

To measure landscape metrics, I created “scenario landscape” maps at 100-year intervals of each replicate of each scenario, in which map patches (classes) were defined as a combination of one of eight vegetation types, one of five quadratic mean diameter classes, and one of four forest canopy percent cover classes (see class definitions in Table 4). Note that although FireBGCv2 divides patches without ever merging them, patches were merged during this processing if adjacent patches had the same vegetation type, cover class, and diameter class. These maps were used to calculate compositional metrics for each scenario landscape, and were input to FRAGSTATS (version 4.0) for calculation of landscape configuration metrics.

NMS ordination of data from all replicates was performed in PC-ORD 6 (McCune and Mefford 2011). The distance measure used was Sorensen (Bray-Curtis), and a general proportional relativization was applied to the data matrix to equalize the weight of all variables. Rank-transformed multi-response permutation procedure (MRPP) was performed to determine whether temperature, precipitation, and fire levels formed significantly differentiated groups based on the test statistic A, which is the chance-corrected within-group agreement, and the probability p that the effect occurred by chance. Mean and quantiles for A were calculated using 1000 bootstrapped samples from the ordination data matrix.

Results

Vegetation composition

Changes in precipitation levels did not strongly affect the relative proportion or geographic distribution of major vegetation types on the landscape, but both temperature and fire suppression did (Figure 8). Changes in area occupied by non-forest (<10% tree canopy cover, which may include woodland, shrubland, and grassland) were especially obvious. Under +3° and +6° C scenarios, non-forest increased in prominence under both fire suppression scenarios, though considerably more so in scenarios with the most fire. For example, with 10% fire suppression and current precipitation levels, non-forest occupied an average of 3%, 13%, and 38% of the landscape at +0°, +3°, and +6°, respectively. With 90% fire suppression, non-forest on average made up less than 2% of the landscape at current temperatures, increasing to 5% and 18% of the landscape under +3° and +6°, respectively.

Conversely, the percentage of the landscape occupied by cool-adapted forest types (e.g., mountain hemlock, cool mixed conifer) was lower with increasing temperature and fire frequency. Again considering only current precipitation scenarios, with 90% fire suppression, cool-adapted forest types decreased from an average of 37% of the landscape at current temperatures to 10% and 1% of the landscape in the +3° and +6° scenarios, respectively. With 10% fire suppression and current precipitation, cool-adapted forests comprised only an average of 28%, 7%, and less than 1% of the landscape under +0°, +3°, and +6° C, respectively. More fire also resulted in considerably less warm moist conifer for a given level of precipitation, especially at +3° and +6°.

The geographic distribution of vegetation types shifted under future climate scenarios (Figure 9, Figure 10). With 90% fire suppression (Figure 9), a 3° increase in temperature caused cool-adapted conifer types to retreat to the highest elevations, while moist mixed conifer (largely fir- or spruce-dominated) moved in behind them, and ponderosa pine types expanded in lower elevations. Under a 6° temperature increase and continued 90% fire suppression, moist mixed conifer shifted upslope; non-forest replaced some conifer forest in low elevations, and cool-adapted conifers were restricted to the highest summit on the landscape. With only 10% of fires suppressed (Figure 10), non-forest was more prominent than under 90% suppression in both +3° and +6° scenarios. In the 10% suppression/ +6° scenarios, low elevations were largely overtaken by non-forest, interspersed with isolated patches of ponderosa pine forest types.

By shifting, vegetation types retained environmental conditions to which they are adapted: for example, accumulated growing season water stress in each vegetation type at year 500 was relatively unaffected by climate scenario (Figure 11). In all scenarios, water stress was highest in drought-tolerant dry forest types such as ponderosa pine and lodgepole pine. These types were also more sensitive to temperature and fire effects than cool-adapted vegetation types, with higher temperatures and more fire generally leading to less growing season water stress. This suggests that frequent fire and higher temperatures lessened stress on remaining trees by limiting recruitment and understory growth, reducing competition. Non-forested areas, with the fewest trees, generally had the least water stress, but was most senstive to climate and fire scenarios.

The percent of the landscape in different forest structural stages was also affected by temperature and fire suppression (Figure 12). In all scenarios, the structural stage with the most area was “large”, or 50-100 cm diameter at breast height (DBH), while the three smallest categories (seedling, sapling, pole, all under 23 cm DBH) each occupied at most 20% of the landscape. Both warming and higher fire frequency tended to diversify the distribution of structural stages. With 3° of warming, these three smallest stages together occupied ~6% and 13% of the landscape with 90% and 10% fire suppression, respectively. With 6° of warming, these values increased to ~21% and 39% with 90% and 10% fire suppression, respectively. In all climate scenarios, less fire suppression also resulted in slightly more area occupied by the largest, oldest structural stage (>100cm DBH), but a temperature increase of 6° dramatically reduced the prominence of this stage relative to the +0° and +3° scenarios under both fire-suppression scenarios.

The impact of fire was also evident in the geographic arrangement of structural stages on the landscape (Figure 13 and Figure 14). More fire created a more highly divided landscape, with a more evenly distributed arrangement of structural stages. However, nearly all increases in smaller structural stages due to fire occurred at relatively low elevations. Since low-elevation areas were also where the majority of the largest structural stage occurred with more fire, it is apparent that with 6° of warming, the largest structural stage was replaced by the smallest stages.

Landscape dynamics over time

Variation in scenario outcomes was captured by two axes in the NMS ordination (Figure 15) that cumulatively explained 97.3% of variation in the dataset (Table 5). Based on rank-transformed multi-response permutation procedure (MRPP), scenarios that differed only in precipitation levels were not more homogeneous than would be expected by chance ($p>0.05$). Groups defined by each level of temperature and fire suppression were significantly more homogeneous than expected by chance, with mean A = 0.40 (25th and 75th quantiles for A = 0.39 and 0.42, respectively) for temperature groups, and mean A = 0.15 (25th and 75th quantiles for A = 0.13 and 0.16, respectively) for fire suppression groups; p was always <0.0001. Variation among ordination scores for replicates of a given scenario was generally higher for scenarios with 10% fire suppression (high fire frequency) and tended to increase over time (Figure 16). In general, there was considerably more variation along Axis 2 than along Axis 1.

In the NMS ordination, Axes 1 and 2 represent gradients in landscape characteristics. Examination of variable correlations to the axes (Table 5) suggests that Axis 1 generally encompasses a gradient in landscape vegetation composition. Landscapes to the left on Axis 1 have larger areas dominated by non-forest or small regenerating trees, and consequently have more area with low tree canopy cover, as well as a lower average basal area per hectare and more numerous snags per hectare. By comparison, landscapes to the right on Axis 1 have relatively little area dominated by shrubs, but more area in medium-sized trees and cool-adapted vegetation types such as mountain hemlock. These landscapes have larger areas of high percent tree canopy cover, more basal area per hectare, and fewer snags.

Axis 2 corresponds to a gradient in landscape structure and heterogeneity. Landscapes nearer the bottom of Axis 2 have more area in which quadratic mean diameter falls into the largest size classes, and fewer trees per hectare; in other words, they tend to have fewer, larger trees than landscapes nearer the top of Axis 2. Landscapes near the bottom of Axis 2 are also more heterogeneous than those near the top: they are more extensively divided into a wider variety of vegetation types. Near the bottom end of Axis 2, patches of vegetation types tend to be smaller and clumped closer to other patches of the same type, but also show a wider variation in patch size and distances between patches of the same vegetation type. Landscapes near the top of Axis 2 are comparatively uniform, arranged in fewer, larger, more contiguous patches that tend to be further apart from other patches of the same type but more regularly spaced on the landscape.

When considering the ordination results, it should be noted that changes in temperature and precipitation acted over the first 100 years of climate change scenarios, and then were held constant, and that during these years the model was also stabilizing; for these reasons the difference between year 1 and year 100 for each scenario should not be over-interpreted. For these and other modeled responses, it is more informative to compare each climate change scenario to results for the current temperature/ current precipitation/ 90% fire suppression scenario, which best represents a scenario of “no further change”, i.e. an extension of current conditions (open blue circles in Figure 15). Similarly, the current temperature/ current precipitation/ 10% fire suppression scenario roughly correlates to a scenario of near-historical fire frequency and no further warming.

The position of scenario trajectories relative to the ordination axes illustrates the progression of scenarios over time in terms of the axis variables (Figure 15). Under the current climate/ 90% fire suppression scenario (“no further change”), the landscape became somewhat more heterogeneous over time and shifted slightly away from domination by cool, closed forest, but did not shift dramatically. With an increase of 3° and sustained 90% fire suppression, the overall pattern and position of the landscape trajectories appeared little changed compared to current temperature trajectories, only shifted toward a state with less closed, cool forest; this reflected a dramatic loss of cool wet conifer forests from the western high elevations of the study area, which were nearly entirely replaced by moist warm mixed conifer and lodgepole pine. With 6° of warming, however, the trajectories were quite different, even under sustained high fire suppression: they shifted toward a state with more non-forest and small regenerating trees, but retained a contiguous, homogeneous configuration.

Under 10% fire suppression, with more fire on the landscape, all temperature/ precipitation combinations traveled a greater distance on the second axis over time than they did under 90% fire suppression, suggesting that, as expected, fire shifted the landscape toward a more heterogeneous, fragmented condition. The trajectory of the +0° and +3° scenarios were nearly identical in this respect, separating only because of differences in vegetation composition represented on Axis 1. However, the +6°scenarios were very dissimilar from other trajectories under 10% fire suppression, with composition shifting dramatically toward non-forest and regenerating trees and configuration remaining comparatively homogeneous over time. In fact, all scenarios with +0° and +3° were relatively clustered, with the +6° scenarios notably separate. Also, whereas frequent fire in the +0° and +3° scenarios created a more heterogeneous landscape, all +6° scenarios, including those with more fire, resulted in a relatively homogeneous landscape dominated by large contiguous patches of non-forest and smaller-diameter, dense forests. This suggests that a threshold is present between 3 and 6 degrees of temperature increase, past which the landscape changes dramatically in multiple respects.

Discussion

These simulations demonstrate the potential dual impact of climate change on this landscape, through its direct effect on vegetation and its control over fire regimes. Even with continued fire suppression, the frequency and severity of fires will likely increase in a warming climate (Fried et al. 2004, Miller et al. 2009), a trend which was also simulated here (see Chapter 2 of this thesis). Because even scenarios with 90% fire suppression experienced higher fire frequency with warming, it is impossible to entirely isolate the effects of rising temperature and climate-mediated increases in fire frequency in these simulations. However, comparisons among scenario outcomes are still instructive.

Changes in temperature and fire frequency proved more important than changes in precipitation on this landscape. Although results for different precipitation scenarios often hinted at a trend (for example, wet scenarios on average produced less non-forest and more moist conifer than dry scenarios with the same temperature and fire suppression level), differences were generally too small and variable to be conclusive. This likely reflects several factors. First, changes in precipitation are expected (and simulated here) to be largest for Fall, Winter, and Spring seasons (Mote and Salathé 2010), leaving precipitation during the growing season, which is already minimal, comparatively unaltered. This means that changes in growing season water availability mainly follow from changes in temperature, as warmer summers dry and heat the landscape simultaneously, confounding temperature and precipitation effects. Similarly, although it would be reasonable to expect that changes in precipitation would alter fire regimes via changes in fuel moisture, such an effect was not apparent. The fire season (nearly the same as the growing season) has always been dry, and is likely to remain so; as with direct vegetation effects, climate change will probably affect fire regime mainly by way of temperature, as warmer temperatures dry the landscape earlier and keep it dry longer each year (Westerling et al. 2006). Also, these simulations do not include increased weather variability that is expected to accompany climate change (Mote and Salathé 2010). Fluctuations such as lengthened wet/ dry cycles may have the potential to alter vegetation types beyond what is represented here (Holden et al. 2007).

Potential landscape trajectories

The spatio-temporal ordination of compositional, structural, and configurational metrics highlighted two main aspects of potential trajectories on this landscape. First, the ordination clearly illustrates the degree to which frequent fire shapes landscape pattern and vegetation composition and structure. Second, the ordination was notable for its decisive separation of the +6° scenarios from the +0° and +3° scenarios. From these trajectories, it appears that while even moderate warming altered vegetation composition, 6° of warming pushed the landscape over an ecological threshold that separated it from the other scenarios early in the simulation. The divergence was even more decisive when more fire was allowed to burn.

Ecological thresholds, or "tipping points", are conditions under which small additional changes may produce large ecological effects, potentially shifting a system to a new state (Groffman et al. 2006). Because they depend on complex, non-linear ecosystem responses, they are difficult to predict, and often are only recognized when they have already been passed—at which point new conditions may be self-sustaining and the changes practically irreversible (Groffman et al. 2006, Stafford et al. 2010). Bachelet et al. (2001) identified a potential warming threshold that fell within the this interval (4.5°C), past which drought stress may become high enough in the US that forest processes would be altered and productivity and landscape carbon storage could decline, an outcome that also occurred here under 6° of warming (Appendix Table A4). On this landscape, the threshold between 3° and 6° of warming represents the potential for dramatic changes in forest composition, structure, and function.

Vegetation composition and structure—comparison to historical conditions

Simulated vegetation composition of forests under the current temperature scenarios with 10% fire suppression generally agreed with estimated composition of pre-Euroamerican East Cascades forests, suggesting that this scenario replicated historical forest composition fairly well, although this was not its purpose. In these current temperature/ high-fire scenarios, ponderosa pine and warm dry conifer occupied approximately 35% of the landscape, similar to the 39% found by Kennedy and Wimberly (2009) for the portion of Deschutes National Forest covered by the Northwest Forest Plan (which includes this landscape); Agee (2003) found that these forests likely made up ~37% of a Washington East Cascades landscape. Approximately

13% of my simulated historical landscape was dominated by warm moist conifer, comparable to historical estimates that it comprised 11% (Kennedy and Wimberly 2009) or 18% (Agee 2003) of similar landscapes. Approximately 18% of current temperature/ high-fire-frequency scenario landscapes were dominated by lodgepole pine, a value fairly similar to the 10% found by Kennedy and Wimberly (2009; lodgepole pine was not a major landscape component in the 2003 Agee study). Cool-adapted subalpine conifer forests—dominated by mountain hemlock, silver fir, and subalpine fir—historically made up approximately 30-40% of East Cascades landscapes (Hessburg et al. 2000, Agee 2003, Kennedy and Wimberly 2009), matching the ~37% simulated here. Parkland, shrubland, and grassland likely made up approximately 2% to 18% of historical East Cascades landscapes (Hessburg et al. 2000, Agee 2003, Kennedy and Wimberly 2009), a range that was also captured in these scenarios.

Before East Cascades dry forests were homogenized by Euroamerican management, they formed a mosaic of diverse forest types and structures (Hessburg et al. 2005, Naficy et al. 2010), conditions that were best replicated in these simulations by scenarios with high fire frequency. The wide variety of structural categories used in research makes them difficult to compare across studies, but estimates of the percentage of East Cascades landscapes occupied by seedling, sapling, and “early successional” structural types range from ~7% to 40% (Agee 2003, Kennedy and Wimberly 2009). Although this is a wide range, my simulations of early structural stages under current climate barely reached 4% even with only 10% of fires suppressed. In the same current climate/ high-fire-frequency scenarios, mature and large (10-50cm DBH) forests occupied approximately 69% of the landscape, higher than the estimated historical range of 35-64% (Hessburg et al. 2000, Agee 2003, Kennedy and Wimberly 2009). The largest, oldest structural stages occupied ~27% of the simulated landscape, comparable to Agee (2003; 20-52%) and Kennedy and Wimberly (2009; ~25%). Based on these previous studies, historical forest structure is best represented in these simulations by the +3°/ high-fire-frequency scenarios. This may be due to the fact that simulated high-fire-frequency scenarios still permit only 90% of fires to burn. Increasing fire frequency that accompanied moderate warming scenarios may therefore more accurately reflect historical fire regimes and related distributions of forest successional stages.

Response of vegetation to warming and fire suppression

Comparison among warming scenarios suggests that while a combination of fire and temperature drove changes in the relative prominance of vegetation types, geographic shifts of vegetation types were driven mainly by temperature. As expected, ponderosa pine and warm mixed conifer forests moved upslope in response to increases in temperature. This upward shift in elevation has been projected previously for this and other ecosystems (e.g. Bachelet et al. 2001, Rehfeldt et al. 2006). Because they were able to shift upslope, the proportion of warm-adapted forest types on the simulated landscape remained relatively unaffected by projected increases in temperature, with only a few warm forest types experiencing minor changes in dominance on the landscape. Unsurprisingly, more fire tended to decrease moist conifer forest relative to dry forest types, as fire reduced the retention of fire-intolerant trees like spruce and true firs. Lodgepole pine, while favored by fire, tended to decrease in prominence with warmer temperatures, potentially reflecting the loss of cold environments to which it is better adapted than other species (Coops and Waring 2011).

As a result of the upslope migration of warm-adapted forest types, simulated forest composition at high elevations underwent dramatic shifts in response to warming. With only 3° of warming, cool conifer forests were reduced from 37% to approximately 10% of the landscape, and disappeared almost entirely under 6° of warming. Lenihan et al. (2003, 2008b) projected a similar trend for subalpine communities in California and across the United States, and simulations of the Northwest and of Oregon ecoregions by Rehfeldt et al. (2006) and Busing et al. (2007) also found that climate warming spurred upslope migration of mesic species, constricting the range of subalpine forest types. In California's central Sierra Nevada, comparisons between 1930s and 1990s vegetation composition data suggest that the loss of high-elevation conifer forest types is already underway (Thorne et al. 2008).

The simulated occurrence of non-forest (woodland, shrubland, or grassland) fell within likely historical levels for most scenarios, excepting the hottest scenarios (+6°) with only 10% fire suppression. In these hot/ high-fire scenarios, non-forest expanded to nearly 40% of the landscape, a condition probably not representative of historical conditions (Kennedy and Wimberly 2009). Past field studies (Savage and Mast 2005) and modeling studies (Lenihan et al. 2008b, Westerling et al. 2011) have suggested that repeated burning, which may occur more often in a warmer climate, can shift landscapes to new physiognomic states, converting forests to woodland, shrubland, or grassland; when fires were allowed to burn in this study, the ecological threshold for such a shift at low elevations appeared to occur between 3 and 6 degrees of temperature increase.

Nonetheless, the patterns of forest structure indicate that fire and increasing temperatures are likely to increase the diversity and prominence of younger forest on the landscape, especially in lower, more frequently burned areas. Such a trend would be beneficial to wildlife species that rely on early seral forest for forage or shelter, and to species like the black-backed woodpecker (*Picoides arcticus*), which responds positively to the presence of post-fire vegetation structure (Murphy and Lehnhausen 1998).

For species like the northern spotted owl (*Strix occidentalis* caurina) that rely on multistoried old forest structure, however, these simulations paint a more nuanced picture. Climate change and related changes in fire regimes are widely expected to negatively impact old-forest-associated species in dry forests, where fire suppression has created dense, multi-storied forest structure. This forest structure was historically rare in dry forests, and is now highly vulnerable to wildfire (McKenzie et al. 2004, Spies et al. 2006, Ager et al. 2007, Kennedy and Wimberly 2009). In this study, area occupied by the oldest structural stage remained unchanged or even increased slightly with only moderate warming. The additional fire in scenarios with 10% fire suppression also appeared to slightly increase the relative area dominated by the largest trees, suggesting that fire promoted large trees by clearing competing undergrowth and small trees. However, the highly fragmented landscape that arises with increasing fire frequency may not provide old patches of sufficient size or vertical complexity for habitat, or may not support necessary prey species (Spies et al. 2006). For these reasons, the presence of old forest is not sufficient to determine habitat suitability for these species. Under the hottest scenario simulated here, the area occupied by the oldest forest decreased dramatically, replaced by non-forest; such a condition would likely offer little habitat for old-forest-associated species.

Conclusions and implications

This study and many others agree that climate change is likely to dramatically impact dry forested landscapes. Warming temperatures and more frequent fire altered vegetation composition, regardless of fire management. Area dominated by cool wet conifer forest decreased, while the extent of non-forest, ponderosa pine, and warm mixed conifer forest increased or remained constant, climbing in elevation to follow environmental envelopes. Even with little fire, 6° of temperature increase was sufficient to nearly eliminate mountain hemlock and cool wet conifer forests. With more fires burning, these effects were amplified for each potential temperature scenario, and forest structure and configuration began to shift away from current large homogeneous stands toward a more fragmented and diverse structural state.

The likelihood that changes such as those presented here actually occur will depend on the magnitude of real changes in climate and on implemented land management strategies, but these results represent a range of possibilities that could suggest alternative paths for management and research. For example, although restoration of open, park-like stands of ponderosa pine and dry conifer may be desirable for safety, recreation, and control of fire behavior, it may prove difficult or impossible to maintain such stands in their present locations as warming temperatures shift the geographic location of suitable environments and potentially replace them with shrubland and juniper from the East. Reduction in area of subalpine forest types may be unavoidable with warming, and would dramatically change the face of the landscape, in addition to constricting habitat for plant and animal species that live there. Because these forests have relatively few commercially important tree species, they are comparatively understudied, and their loss may have unforeseen consequences.

Some studies have suggested that novel future climatic conditions are likely to lead to novel or “no-analog” plant communities, in which combinations or abundances of species arise that are previously unknown in a biome (Hobbs et al. 2006, Williams and Jackson 2007, Williams et al. 2007). Since many models simulate vegetation as consolidated vegetation “types”, rather than as species responding individually to environmental factors, our ability to predict future novel communities is limited, especially under future climate conditions (Williams and Jackson 2007), which may limit our ability to plan appropriate management strategies. FireBGCv2’s species-specific physiological and phenological parameterization makes it an appropriate model to use in explorations of potential no-analog communities. In this study, however, little evidence of such communities appeared, and species were therefore consolidated into vegetation types for simplicity. The low overall number of simulated tree species may have limited the potential for no-analog conditions, or species parameterizations may not have been accurate enough for trends to emerge. Additionally, the simplistic, non-species-specific parameterization of undergrowth eliminated the possibility of analyzing unsynchronized shifts among overstory and undergrowth species. Further investigation of potential no-analog communities in this area might provide more insight into possible ecological or management implications.

Even if the future landscape were to fall within its historical range of variability, changes such as conversion of forest to shrubland or grassland may be undesirable for ecological, economic, or social reasons. Loss of low-elevation forests would decrease the potential for carbon sequestration and storage in those areas (Bachelet et al. 2001). In these simulations, the hottest temperatures and low fire suppression resulted in 40-50% less total carbon per unit area on the landscape than under current conditions (Appendix Table A4). Changes to habitat may further constrict ranges of threatened species and complicate interactions between habitat management and fuels and fire management (Lee and Irwin 2005, Kennedy and Wimberly 2009).

A changing landscape, and the altered management plans that would likely accompany it, may also increase friction between what is ecologically possible and what is socially acceptable (Duncan et al. 2010). Social acceptability may prove to be a particular challenge on this landscape, because it includes wildlife habitat, it is part of an important watershed, and it is a popular recreation area in which many people may feel emotionally invested. For example, if timber harvest goals become difficult to sustain at lower elevations, changes to harvest plans might be required, and could present conflicts with other management considerations like recreation and wildlife habitat. Additionally, if timber resources must move upslope to follow productivity, timber harvesting may be subject to growing costs and conflicts associated with the need for new roads, steeper and more rugged harvest units, and longer transport distances. Changes on the landscape such as those projected here will require managers to adapt and adjust continually to ensure that all management goals can be addressed. Monitoring for and adapting to changes as they occur will be necessary to ensure that strategies are not destined to fail because of fundamental conflicts with the environmentally determined trajectory of the landscape.

LIMITATIONS

In most modeling studies, including this one, the value of the simulations lies less in the absolute numbers they produce than in comparison among potential scenarios, and the information those comparisons provide regarding potential ecosystem function and development. Like all modeling projects, this study is limited by the data used for parameterization and by the assumptions of the modeling platform.

For example, this parameterization of FireBGCv2 omits insects and pathogens, such as mountain pine beetle (*Dendroctonus ponderosae*), which have wide-ranging effects on East Cascades landscapes and may interact unpredictably with changes in climate and fire regimes (McKenzie et al. 2004, Littell et al. 2010, Hicke et al. 2012); potential management actions like logging, thinning, and prescribed burning are also excluded.

Each model also has its own quirks; for example, although fire behavior in FireBGCv2 responds to available fuel and fuel conditions, simulated fire spread relies only on vectors of wind and slope. Consequently, it was difficult to model historical fire regimes on this landscape, where extreme temperature and fuel moisture gradients can exert considerable control over fire spread. Fuel moisture conditions are simulated at a coarse (site) level, so stand-level fuel moisture variability arising from differences in aspect and slope are largely unaccounted for. Simulated wind speeds only vary within a narrow window, meaning that extreme fire weather events are not simulated.

Tree growth algorithms in FireBGCv2 are species specific, but four minor species in this study (sugar pine, incense-cedar, noble fir, and giant chinquapin) were not present in the model due to a lack of parameters and algorithms, so algorithms for similar species were used as surrogates. Also, species presence in the model is limited to those species included at the beginning of the simulation; the model cannot simulate the immigration of species from adjacent systems. In particular, it is possible that higher temperatures would lead to juniper encroachment from the east, especially with continued fire suppression. Finally, shrub species are not individually parameterized in this implementation of FireBGCv2. Each shrub species interacts differently with fire according to its physiology and life history (e.g. flammability, sprouting ability), and characteristics of shrub communities can control local fire regimes (Keeley et al. 2008), but these dynamics are not captured here.

More generally, the use of patches to represent biophysical settings and community types in maps and models—as in this study—is widespread, but patches and patch-based metrics are increasingly viewed as inadequate representations of ecosystem attributes that naturally occur as gradients (Cushman et al. 2008b, McGarigal et al. 2009).

FIGURES AND TABLES

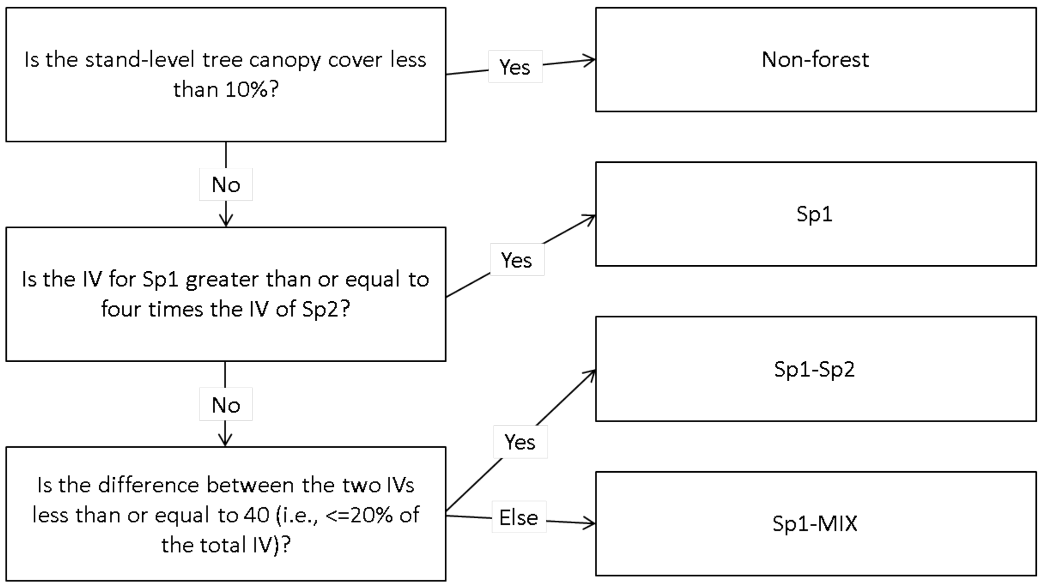

Figure 7. Decision tree for assigning preliminary forest types. See text for explanation of abbreviations.

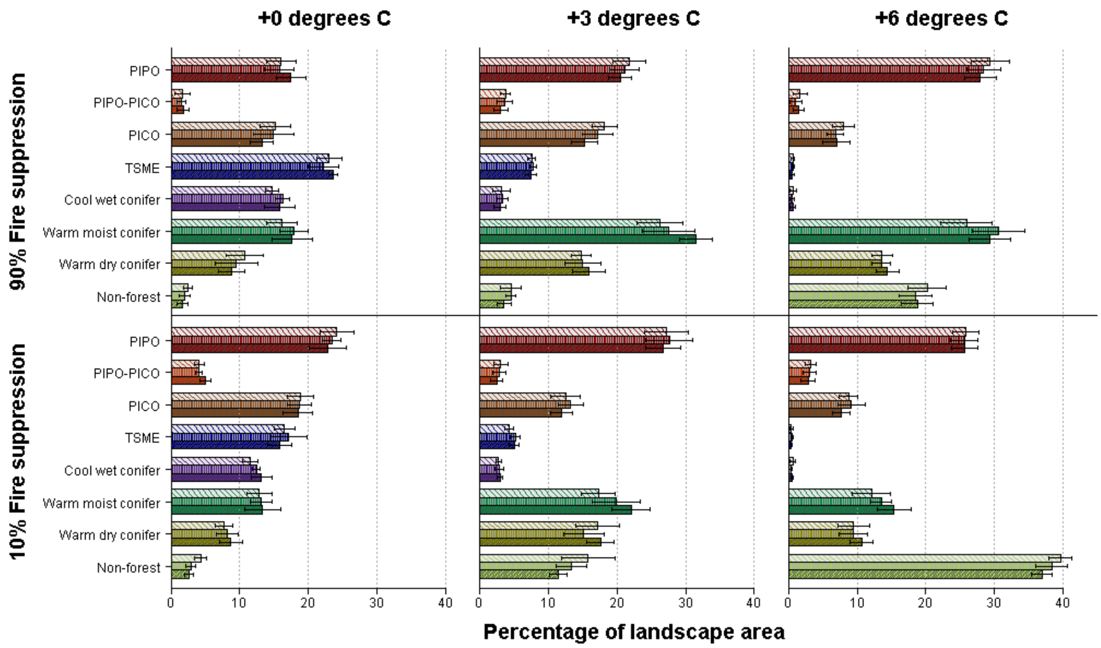

Figure 8. Mean percentage of the landscape occupied by vegetation types at year 500. Whiskers are standard deviation. In each group of three bars, the bottom bar is the wet scenario, middle bar is the current precipitation scenario, and top bar is the dry scenario. PIPO = ponderosa pine; PICO = lodgepole pine; TSME = mountain hemlock.

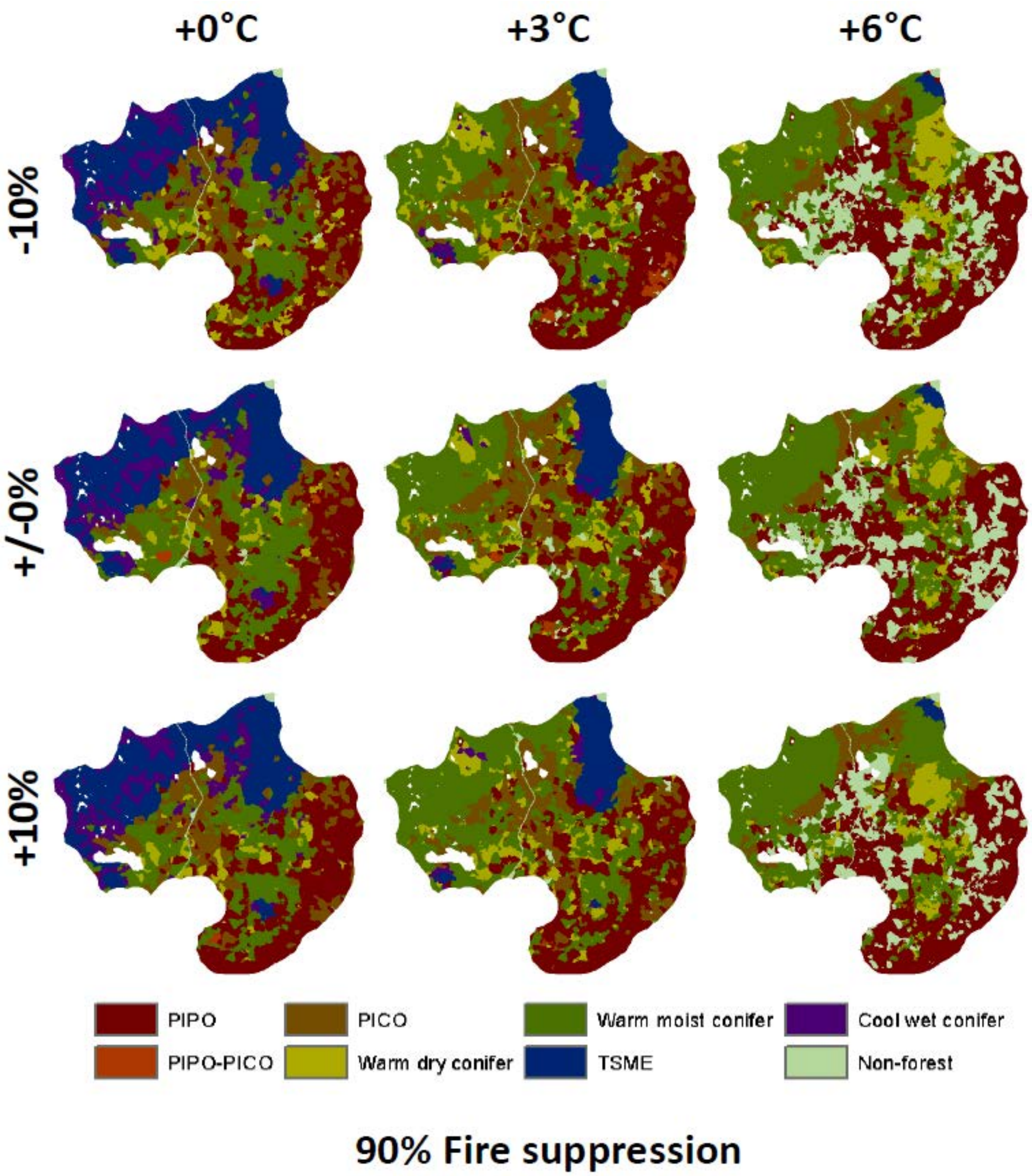

Figure 9. Geographic distribution of forest types under potential climate scenarios and 90% fire suppression at year 500. PIPO = ponderosa pine; PICO = lodgepole pine; TSME = mountain hemlock.

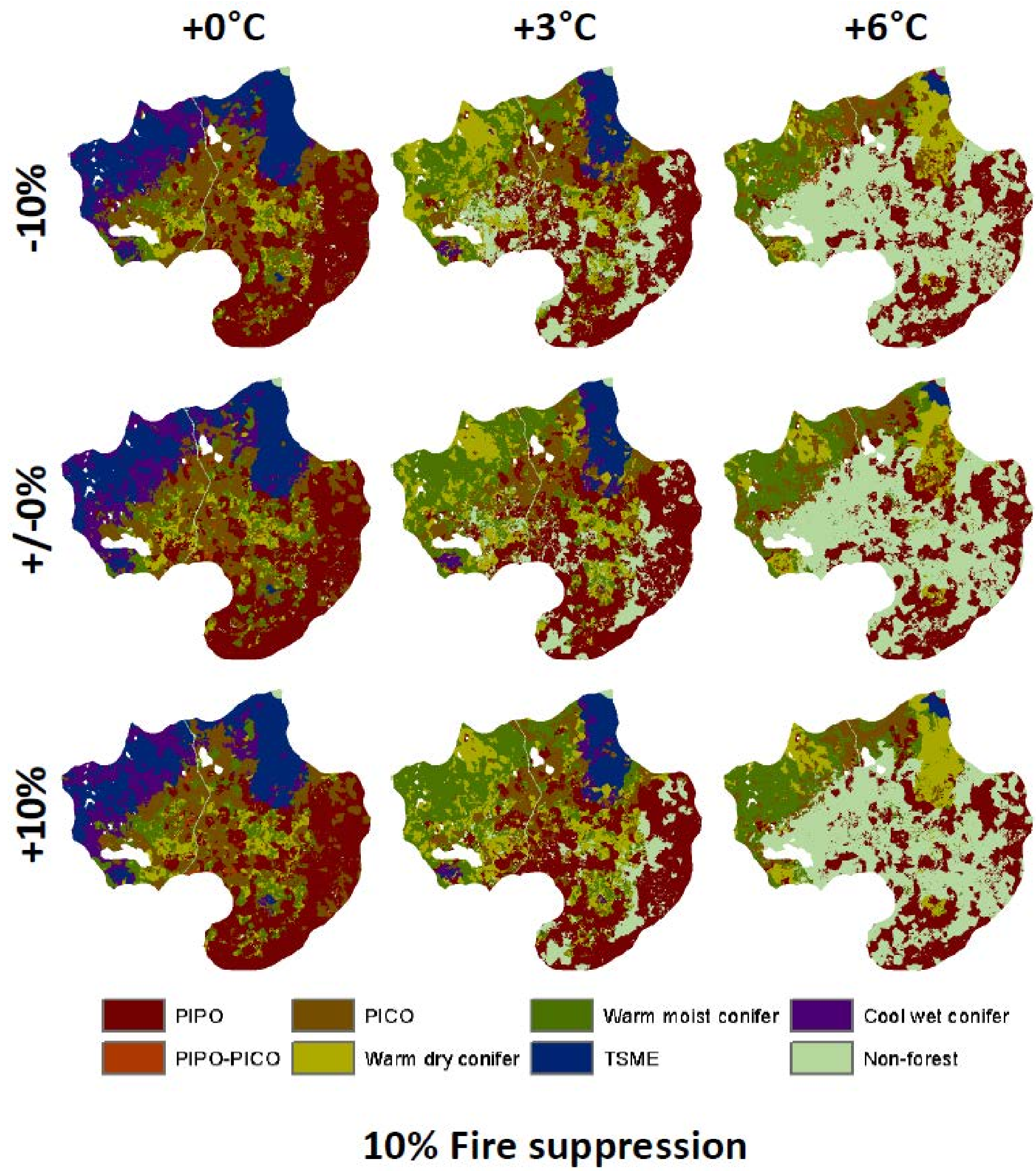

Figure 10. Geographic distribution of forest types under potential climate scenario and 10% fire suppression at year 500. PIPO = ponderosa pine; PICO = lodgepole pine; TSME = mountain hemlock.

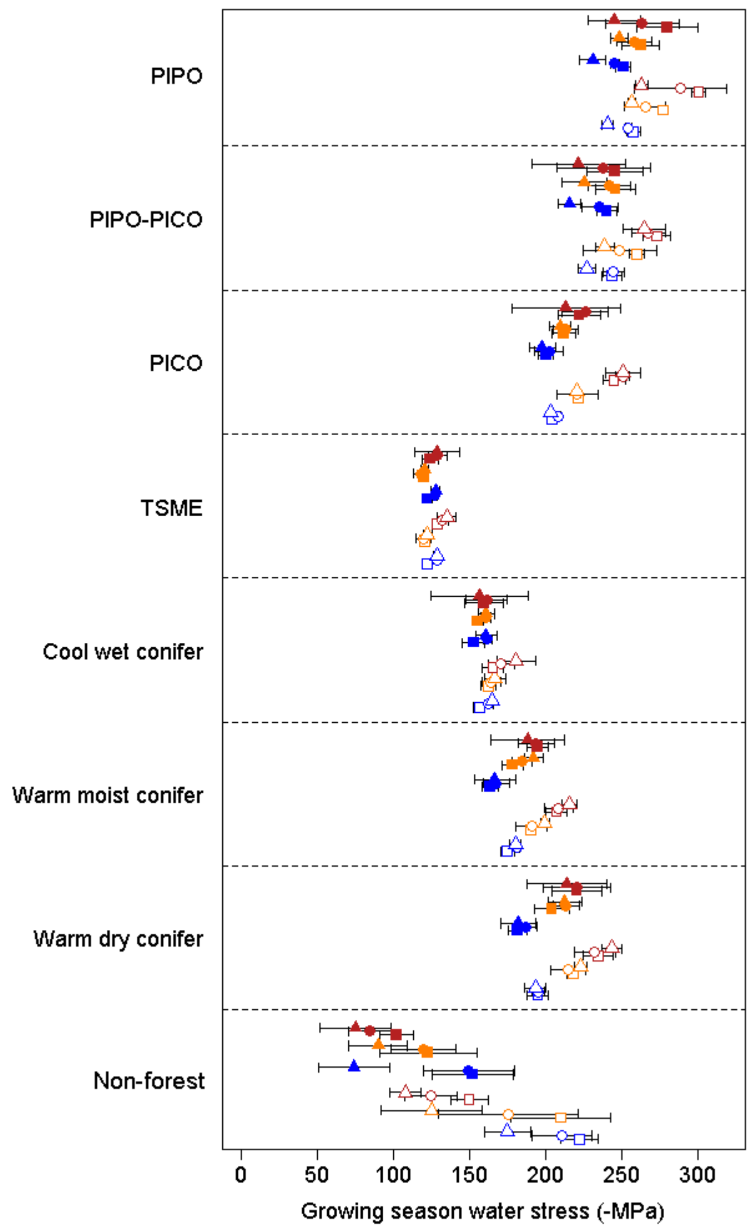

Figure 11. Accumulated mean growing season water stress for vegetation types in each scenario at year 500. Whiskers are standard deviation. Open symbols: 90% fire suppression; closed symbols: 10% fire suppression. Blue: +0°°; orange: +3°; red: +6°. Squares: +10% precipitation; circles: no change in precipitation; triangles: -10% precipitation. PIPO = ponderosa pine; PICO = lodgepole pine; TSME = mountain hemlock.

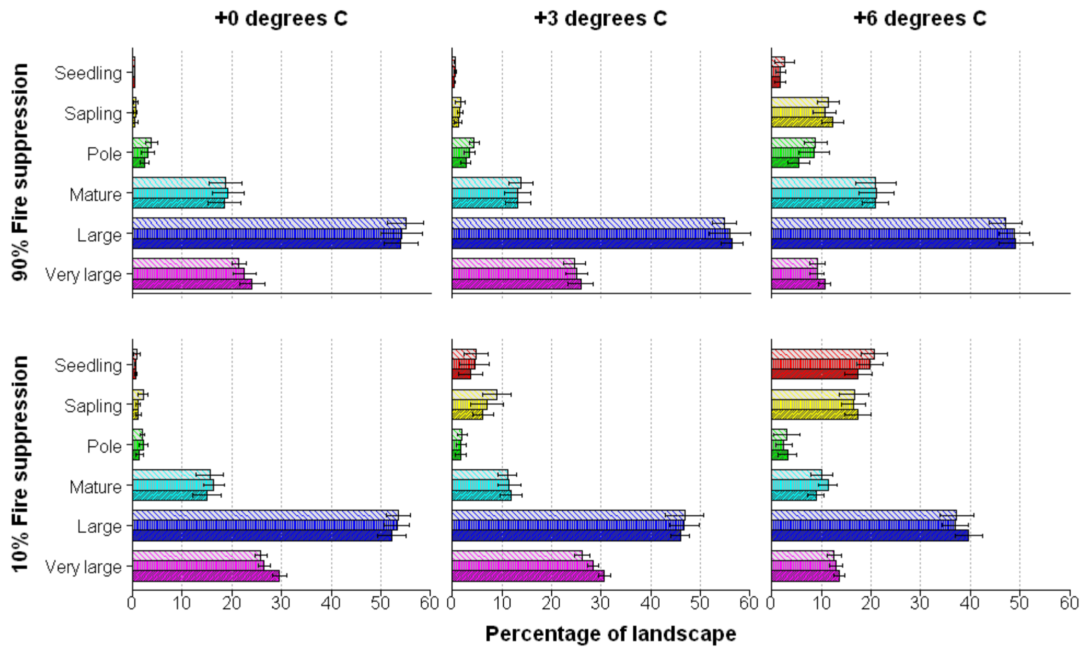

Figure 12. Mean percentage of the landscape occupied by structural stages at year 500. Whiskers are standard deviation. In each group of three bars, the bottom bar is the wet scenario, middle bar is the current precipitation scenario, and top bar is the dry scenario. Seedling: <2cm DBH; sapling: 2-10cm DBH; pole: 10-23cm DBH; mature: 23-50cm DBH; large: 50-100cm DBH; very large: >100cm DBH.

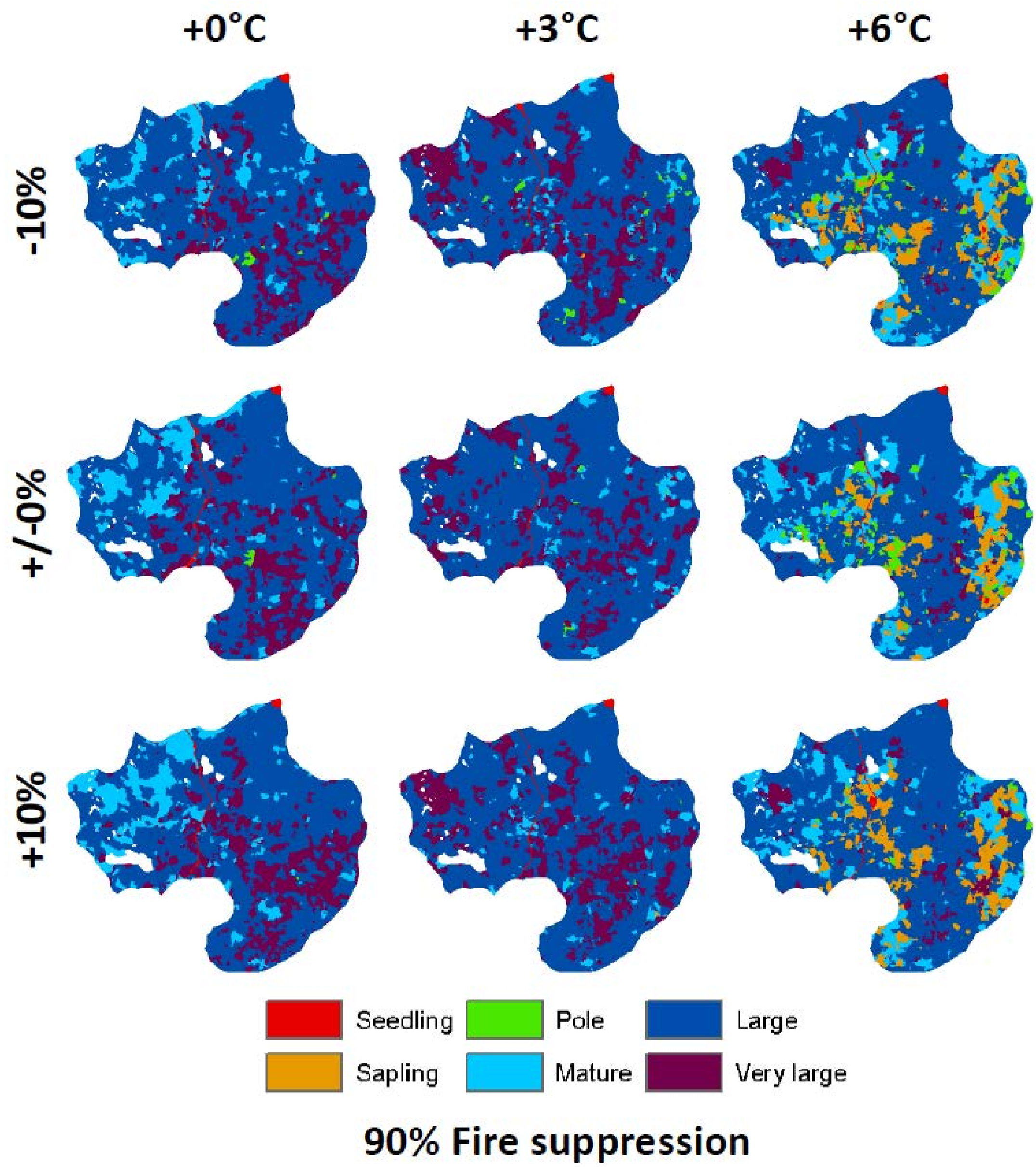

Figure 13. Geographic distribution of vegetation structural stages under potential climate scenarios and 90% fire suppression. Seedling: <2cm diameter at breast height (DBH); sapling: 2-10cm DBH; pole: 10-23cm DBH; mature: 23-50cm DBH; large: 50-100cm DBH; very large: >100cm DBH.

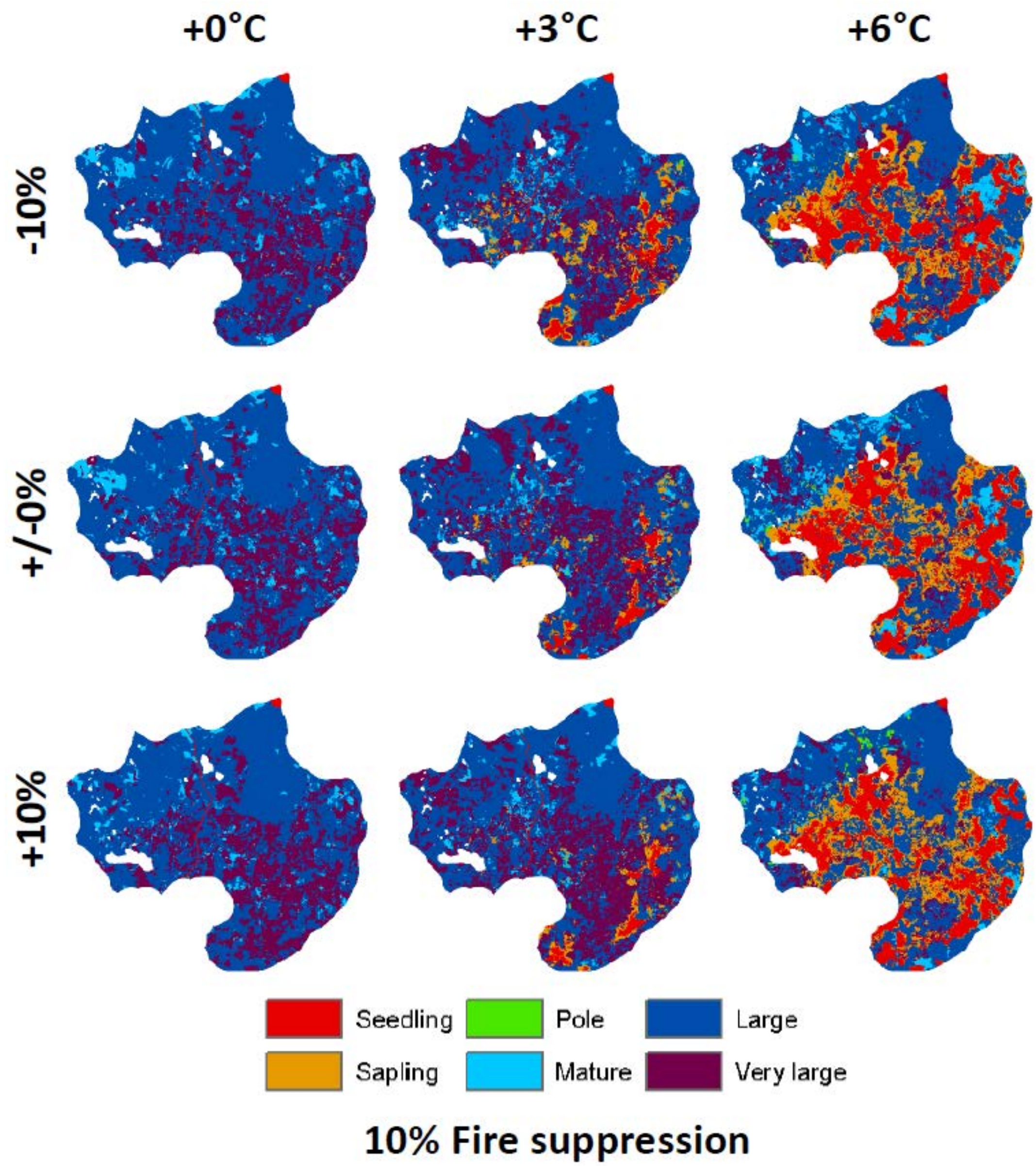

Figure 14. Geographic distribution of vegetation structural stages under potential climate scenarios and 10% fire suppression. Seedling: <2cm diameter at breast height (DBH); sapling: 2-10cm DBH; pole: 10-23cm DBH; mature: 23-50cm DBH; large: 50-100cm DBH; very large: >100cm DBH.

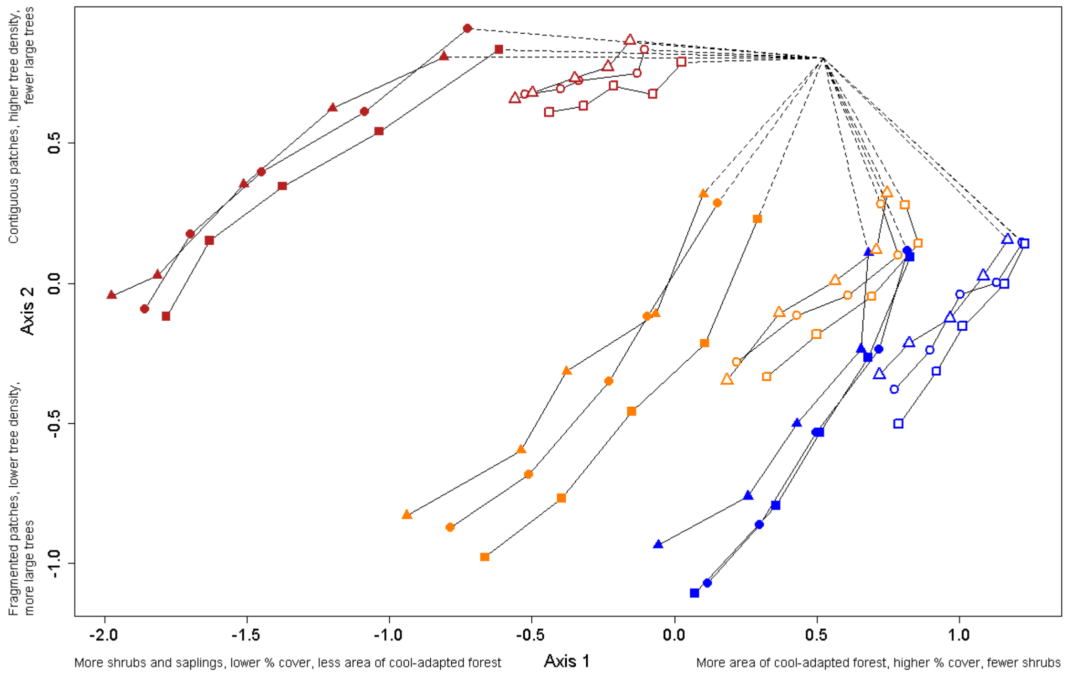

Figure 15. Landscape trajectories under potential future climate change and fire suppression scenarios. Median scores are plotted. Dotted lines represent the first 100 years for each scenario; solid lines trace landscape trajectories for climate and fire scenarios, with symbols at 100-year intervals. Open symbols: 90% fire suppression; closed symbols: 10% fire suppression. Blue: +0°; orange: +3°; red: +6°. Squares: +10% precipitation; circles: no change in precipitation; triangles: -10% precipitation. See text for explanation of axes.

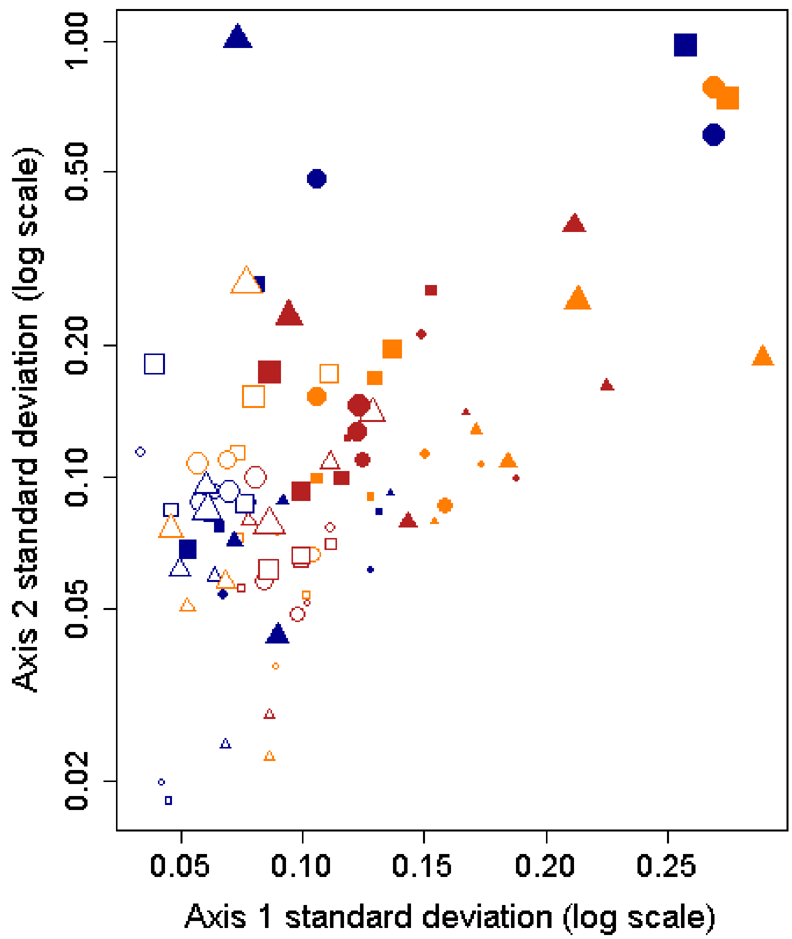

Figure 16. Standard deviation of ordination scores for each scenario. Plotted on a log scale for clarity. Open symbols: 90% fire suppression; closed symbols: 10% fire suppression. Blue: +0°; orange: +3°; red: +6°. Squares: +10% precipitation; circles: no change in precipitation; triangles: -10% precipitation; symbol size increases with each 100-year timestep (100-500).

Table 4. Landscape metric variables included in ordination analysis, with variable codes.

Variable	Variable code
Percentage of landscape area in each of five quadratic mean diameter classes:	
• 0 cm (non-forested)	QMD0
• <12.7 cm (<5 inches)	QMD1
• 12.7-38.1 cm (5-15 inches)	QMD2
• 38.1-76.2 cm (15-30 inches)	QMD3
• >76.2 cm (>30 inches)	QMD4
Percentage of landscape in each of four canopy percent cover classes:	
• <10%	COV0
• 10-40%	COV1
• 40-60%	COV2
• >60%	COV3
Percentage of landscape in each of four general vegetation groups:	
• Non-forested	VEG1
• Pine (ponderosa pine, lodgepole pine, ponderosa-lodgepole mix)	VEG2
• Warm mixed conifer (warm dry conifer, warm moist conifer)	VEG3
• Cool mixed conifer (mountain hemlock, cool wet conifer)	VEG4
Landscape basal area per hectare (m^2/ha)	BAha
Landscape number of trees per hectare	NTha
Landscape number of snags per hectare	NSha
Sum of landscape shrub and herb biomass (kg/m^2)	ShHbio
Patch richness*	PR
Patch density*	PD
Patch size coefficient of variation*	AREA_CV
Edge density*	ED
Contagion*	CONTAG
Area-weighed mean fractal dimension*	FRAC_AM
Interspersion and juxtaposition*	IJI
Simpson's evenness index*	SIEI
Simpson's diversity index*	SIDI
Area-weighted mean Euclidean nearest-neighbor distance*	ENN_AM
Euclidean nearest-neighbor coefficient of variation*	ENN_CV

* Definitions of landscape metrics can be found in McGarigal and Marks (1995).

Table 5. Nonmetric multidimensional scaling ordination axes showing variables with strongest correlations for each axis. See Table 4 for variable codes.

Axis	Percent variance explained (cumulative)	Variable	Correlation with axis
Axis 1	74.7% (74.7%)	QMD2	0.959
		COV0	-0.937
		VEG1	-0.937
		COV3	0.912
		BAha	0.878
		QMD1	-0.810
		QMD0	-0.786
		COV1	-0.784
		NSha	-0.767
		VEG4	0.754
Axis 2	22.6% (97.3%)	QMD3	-0.826
		QMD4	-0.686
		PR	-0.578
		PD	-0.578
		ED	-0.534
		NTha	0.514

CHAPTER 4—CONCLUSION

This project posited a variety of climate futures, and explored potential effects of those climates on vegetation and fire in a popular, ecologically diverse East Cascades landscape. The results demonstrate the ability of spatial landscape models like FireBGCv2 to help us visualize divergent outcomes and consequences of changes whose magnitude are as yet unknown. They also suggest that such changes have the potential to dramatically alter the current landscape.

It seems clear that fire, after almost a century of near-exclusion, is poised to regain its place as a major force for disturbance in the East Cascades. Significant fire events like the Davis and B&B fires suggest that this transition is already under way, needing only time for the proper combination of weather and fire ignitions to occur and provide opportunities for progress. In this study, the combination of heavy, continuous fuel loads and longer, hotter fire seasons resulted in more fire events even with sustained attempts at fire suppression, and those events were high intensity unless fuel loads were diminished by repeated fire and loss of forest productivity. When fire frequency increased sufficiently under warming temperatures, diminishing fuels began to restrain fire intensity; it was noteworthy, however, that sustained fire suppression without increased fire frequency (as in the no-further-change scenario) maintained high-intensity fire on the landscape over time.

FireBGCv2 is a research tool and is not meant to be used in short-term land management decisions, but the long-term outcomes simulated here may still suggest short-term strategies. For example, sustained high-intensity fire with fire suppression suggests that a policy of allowing fires to burn whenever possible (wildland fire use) will decrease the intensity of re-burning fires, lessening future risk to human and ecological resources. In areas where it is unsafe or impractical to allow fires to remove fuels, effective fuel treatments applied at large scales may achieve similar reductions in fire intensity.

Given the small scale at which fuel treatments can usually be applied, however, widespread uncharacteristically high-intensity fires seem inevitable in the near future. Although this could restore heterogeneity to the landscape, and restore a balance of fire-tolerant and fire-intolerant forest types and structures, it may also alter vegetation composition beyond the landscape's historical or socially acceptable range of variability, especially when combined with direct effects of warming on vegetation. Consistently successful fire suppression may be necessary in low elevations to prevent conversion to non-forest.

Even if fire managers succeed in suppressing fire, the landscape may undergo significant changes in response to warming. In this study, higher temperatures led to migration of major forest types on the landscape such that cool subalpine forests were severely reduced under moderate warming and essentially eliminated with severe warming. Meanwhile, non-forest vegetation types displaced ponderosa pine forests in lower elevations, a change that especially reduced the prominence of the largest forest structural stage on the landscape. Such changes would render the landscape quite a different place than it is today.

Taking into account composition, structure, and landscape pattern, overall differences between landscapes simulated with 0 and 3° C of warming were considerably less dramatic than differences between landscapes with 3 and 6° C of warming, especially with frequent fire. This indicates that an ecological threshold may exist in that temperature interval under the precipitation and fire suppression conditions simulated here. Whether this threshold is due to direct effects of warming or to combined effects of warming and climate-mediated changes in fire regime is difficult to determine from these results, but its presence suggests interesting potential future investigations, and also speaks to the need for flexibility and responsiveness in land management strategies.

This study provides a wealth of information on possible effects of climate change on this East Cascades landscape, but it is essentially exploratory. To truly investigate landscape futures, additional research could take better advantage of available modeling tools. Scenarios that include land management activities like thinning and prescribed fire would more accurately reflect development trajectories on this extensively managed landscape. Inclusion of important disturbance vectors like mountain pine beetle (*Dendroctonus ponderosae*) and white pine blister rust (*Cronartium ribicola*) would affect simulated outcomes for both fire and vegetation dynamics. Additional computing resources could allow incorporation of improved fire spread algorithms to better capture the effects of strong moisture gradients on fire regimes in this important transitional zone between maritime and continental climates. Further exploring the balance of potential precipitation changes and potential temperature changes could provide refined insight into future vegetation composition, especially in low-elevation forests that currently survive at an environmental extreme of moisture limitation. Precipitation changes simulated here were not sufficient to overcome moisture deficits due to warming, but precipitation projections are highly uncertain. Precipitation increases sufficient to spur forest productivity have the potential to dramatically alter landscape vegetation and fire processes in ways not represented here.

The statistician George E. P. Box famously noted that “all models are wrong, but some are useful” (Box and Draper 1987). The usefulness of models lies in comparing the glimpses they provide of alternative potential futures, and the way those glimpses may spur additional investigations and reevaluation of management strategies. Current forest management and restoration goals frequently rely on historical conditions to guide strategies, but this study implies that neither current nor historical conditions provide an appropriate reference point for the future of this landscape. While recognizing the value of historical forest condition as a source of information on potential ecological composition and function, the widespread changes in fire and vegetation dynamics simulated in this study suggest that it may be more realistic to strive less for “restoration” and more for landscape conditions that reflect the best available understanding of what is both socially desirable and ecologically possible.

References

Agee, J. K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. Gen. Tech. Rep. GTR-PNW-320, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Agee, J. K. 2003. Historical range of variability in eastern Cascades forests, Washington, USA. Landscape Ecology 18:725-740.

Agee, J. K. and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83-96.

Ager, A. A., M. A. Finney, B. K. Kerns, and H. Maffei. 2007. Modeling wildfire risk to northern spotted owl (Strix occidentalis caurina) habitat in Central Oregon, USA. Forest Ecology and Management 246:45-56.

Albini, F. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. GTR-INT-030, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Bachelet, D., R. P. Neilson, J. M. Lenihan, and R. J. Drapek. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. Ecosystems 4:164-185.

Baker, W. L. 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon’s eastern Cascades. Ecosphere 3.

Bork, J. 1985. Fire history in three vegetation types on the eastern side of the Oregon Cascades. PhD Dissertation. Oregon State University, Corvallis.

Bowman, D., J. K. Balch, P. Artaxo, W. J. Bond, J. M. Carlson, M. A. Cochrane, C. M. D'Antonio, R. S. DeFries, J. C. Doyle, S. P. Harrison, F. H. Johnston, J. E. Keeley, M. A. Krawchuk, C. A. Kull, J. B. Marston, M. A. Moritz, I. C. Prentice, C. I. Roos, A. C. Scott, T. W. Swetnam, G. R. van der Werf, and S. J. Pyne. 2009. Fire in the Earth System. Science 324:481-484.

Box, G. E. P. and N. R. Draper. 1987. Empirical Model-Building and Response Surfaces. John Wiley & Sons, Oxford, England.

Burns, R. M. and B. H. Honkala. 1990. Silvics of North America: Volume 1. Conifers. Agriculture Handbook 654, USDA Forest Service, Washington, D.C.

Busing, R. T., A. M. Solomon, R. B. McKane, and C. A. Burdick. 2007. Forest dynamics in oregon landscapes: Evaluation and application of an individual-based model. Ecological Applications 17:1967-1981.

Chmura, D. J., P. D. Anderson, G. T. Howe, C. A. Harrington, J. E. Halofsky, D. L. Peterson, D. C. Shaw, and J. B. St Clair. 2011. Forest responses to climate change in the northwestern United States: Ecophysiological foundations for adaptive management. Forest Ecology and Management 261:1121-1142.

Coops, N. C., R. A. Hember, and R. H. Waring. 2010. Assessing the impact of current and projected climates on Douglas-Fir productivity in British Columbia, Canada, using a process-based model (3-PG). Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 40:511-524.

Coops, N. C. and R. H. Waring. 2011. A process-based approach to estimate lodgepole pine (Pinus contorta Dougl.) distribution in the Pacific Northwest under climate change. Climatic Change 105:313-328.

Coops, N. C., R. H. Waring, and B. E. Law. 2005. Assessing the past and future distribution and productivity of ponderosa pine in the Pacific Northwest using a process model, 3-PG. Ecological Modelling 183:107-124.

Cushman, S. A. and K. McGarigal. 2007. Multivariate landscape trajectory analysis: An example using simulation modeling of American marten habitat change under four timber harvest scenarios. In: J. A. Bissonette and I. Storch, editors. Temporal dimensions of landscape ecology. Springer Science, New York, NY.

Cushman, S. A., K. McGarigal, and M. C. Neel. 2008a. Parsimony in landscape metrics: Strength, universality, and consistency. Ecological Indicators 8:691-703.

Cushman, S. A., K. S. McKelvey, C. H. Flather, and K. McGarigal. 2008b. Do forest community types provide a sufficient basis to evaluate biological diversity? Frontiers in Ecology and the Environment 6:13-17.

Dillon, G. K., Z. A. Holden, P. Morgan, M. Crimmins, E. K. Heyerdahl, and C. H. Luce. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. Ecosphere 2.

Duncan, S. L., B. C. McComb, and K. N. Johnson. 2010. Integrating Ecological and Social Ranges of Variability in Conservation of Biodiversity: Past, Present, and Future. Ecology and Society 15.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. B. Mickelson, S. Y. Lee, and D. P. Lettenmaier. 2010. Implications of 21st century climate change for the hydrology of Washington State. Climatic Change 102:225-260.

Everett, R. L., R. Schellhaas, D. Keenum, D. Spurbeck, and P. Ohlson. 2000. Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management 129:207-225.

Flannigan, M. D., M. A. Krawchuk, W. J. de Groot, B. M. Wotton, and L. M. Gowman. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire 18:483-507.

Franklin, J. F. and C. T. Dyrness. 1988. Natural Vegetation of Oregon and Washington. Oregon State University Press, Corvallis, OR.

Fried, J. S., M. S. Torn, and E. Mills. 2004. The impact of climate change on wildfire severity: A regional forecast for northern California. Climatic Change 64:169-191.

Gonzalez, P., R. P. Neilson, J. M. Lenihan, and R. J. Drapek. 2010. Global patterns in the vulnerability of ecosystems to vegetation shifts due to climate change. Global Ecology and Biogeography 19:755-768.

Graham, R. T., S. McCaffrey, and T. B. Jain. 2004. Science Basis for Changing Forest Structure to Modify Wildfire Behavior and Severity. RMRS-GTR-120, USDA Forest Service.

Groffman, P., J. Baron, T. Blett, A. Gold, I. Goodman, L. Gunderson, B. Levinson, M. Palmer, H. Paerl, G. Peterson, N. Poff, D. Rejeski, J. Reynolds, M. Turner, K. Weathers, and J. Wiens. 2006. Ecological thresholds: The key to successful environmental management or an important concept with no practical application? Ecosystems 9:1-13.

Haugo, R. D., S. A. Hall, E. M. Gray, P. Gonzalez, and J. D. Bakker. 2010. Influences of climate, fire, grazing, and logging on woody species composition along an elevation gradient in the eastern Cascades, Washington. Forest Ecology and Management 260:2204-2213.

He, H. S., R. E. Keane, and L. R. Iverson. 2008. Forest landscape models, a tool for understanding the effect of the large-scale and long-term landscape processes. Forest Ecology and Management 254:371-374.

Hessburg, P. F. and J. K. Agee. 2003. An environmental narrative of Inland Northwest United States forests, 1800-2000. Forest Ecology and Management 178:23-59.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modem eras. Forest Ecology and Management 211:117-139.

Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management 136:53-83.

Hessl, A. E. 2011. Pathways for climate change effects on fire: Models, data, and uncertainties. Progress in Physical Geography 35:393-407.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2001. Spatial controls of historical fire regimes: A multiscale example from the interior west, USA. Ecology 82:660-678.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. Holocene 12:597-604.

Hicke, J. A., M. C. Johnson, L. H. D. Jane, and H. K. Preisler. 2012. Effects of bark beetle-caused tree mortality on wildfire. Forest Ecology and Management 271:81-90.

Hobbs, R. J., S. Arico, J. Aronson, J. S. Baron, P. Bridgewater, V. A. Cramer, P. R. Epstein, J. J. Ewel, C. A. Klink, A. E. Lugo, D. Norton, D. Ojima, D. M. Richardson, E. W. Sanderson, F. Valladares, M. Vila, R. Zamora, and M. Zobel. 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15:1-7.

Holden, Z. A., P. Morgan, M. A. Crimmins, R. K. Steinhorst, and A. M. S. Smith. 2007. Fire season precipitation variability influences fire extent and severity in a large southwestern wilderness area, United States. Geophysical Research Letters 34.

IPCC. 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, Pachauri, R.K and Reisinger, A. (eds.)]. IPCC, Geneva, Switzerland.

Jordan, G. J., M. J. Fortin, and K. P. Lertzman. 2008. Spatial pattern and persistence of historical fire boundaries in southern interior British Columbia. Environmental and Ecological Statistics 15:523-535.

Keane, R. E., G. J. Cary, I. D. Davies, M. D. Flannigan, R. H. Gardner, S. Lavorel, J. M. Lenihan, C. Li, and T. S. Rupp. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecological Modelling 179:3-27.

Keane, R. E., L. M. Holsinger, and S. D. Pratt. 2006. Simulating historical landscape dynamics using the landscape fire succession model LANDSUM version 4.0. Gen. Tech. Rep. GTR-RMRS-171, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keane, R. E., R. A. Loehman, and L. M. Holsinger. 2011. The FireBGCv2 landscape fire succession model: a research simulation platform for exploring fire and vegetation dynamics. Gen. Tech. Rep. GTR-RMRS-255, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keane, R. E., P. Morgan, and J. D. White. 1999. Temporal patterns of ecosystem processes on simulated landscapes in Glacier National Park, Montana, USA. Landscape Ecology 14:311-329.

Keane, R. E., K. C. Ryan, and S. W. Running. 1996. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model FIRE-BGC. Tree Physiology 16:319-331.

Keeley, J. E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire 18:116-126.

Keeley, J. E., T. Brennan, and A. H. Pfaff. 2008. Fire severity and ecosytem responses following crown fires in California shrublands. Ecological Applications 18:1530-1546.

Kennedy, R. S. H. and M. C. Wimberly. 2009. Historical fire and vegetation dynamics in dry forests of the interior Pacific Northwest, USA, and relationships to Northern Spotted Owl (Strix occidentalis caurina) habitat conservation. Forest Ecology and Management 258:554-566.

Latta, G., H. Temesgen, D. Adams, and T. Barrett. 2010. Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management 259:720-729.

Lee, D. C. and L. L. Irwin. 2005. Assessing risks to spotted owls from forest thinning in fire-adapted forests of the western United States. Forest Ecology and Management 211:191-209.

Lenihan, J. M., D. Bachelet, R. P. Neilson, and R. Drapek. 2008a. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change 87:S215-S230.

Lenihan, J. M., D. Bachelet, R. P. Neilson, and R. Drapek. 2008b. Simulated response of conterminous United States ecosystems to climate change at different levels of fire suppression, CO2 emission rate, and growth response to CO2. Global and Planetary Change 64:16-25.

Lenihan, J. M., R. Drapek, D. Bachelet, and R. P. Neilson. 2003. Climate change effects on vegetation distribution, carbon, and fire in California. Ecological Applications 13:1667-1681.

Leung, L. R., Y. Qian, X. D. Bian, W. M. Washington, J. G. Han, and J. O. Roads. 2004. Mid-century ensemble regional climate change scenarios for the western United States. Climatic Change 62:75-113.

Littell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western U. S. ecoprovinces, 1916-2003. Ecological Applications 19:1003-1021.

Littell, J. S., E. E. Oneil, D. McKenzie, J. A. Hicke, J. A. Lutz, R. A. Norheim, and M. M. Elsner. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change 102:129-158.

Liu, Y. Q., J. Stanturf, and S. Goodrick. 2010. Trends in global wildfire potential in a changing climate. Forest Ecology and Management 259:685-697.

Loehman, R. A., J. A. Clark, and R. E. Keane. 2011. Modeling effects of climate change and fire management on western white pine (*Pinus monticola*) in the northern Rocky Mountains, USA. Forests 2:832-860.

Lutz, J. A., J. W. van Wagtendonk, A. E. Thode, J. D. Miller, and J. F. Franklin. 2009. Climate, lightning ignitions, and fire severity in Yosemite National Park, California, USA. International Journal of Wildland Fire 18:765-774.

McCune, B., J. B. Grace, and D. L. Urban. 2002. Analysis of Ecological Communities. MjM Software, Gleneden Beach, OR, USA.

McCune, B. and M. J. Mefford. 2011. PC-ORD. Multivariate analysis of ecological data. MjM Software, Gleneden Beach, Oregon, USA.

McGarigal, K. and B. J. Marks. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. PNW-GTR-351, USDA Forest Service.

McGarigal, K., S. Tagil, and S. A. Cushman. 2009. Surface metrics: an alternative to patch metrics for the quantification of landscape structure. Landscape Ecology 24:433-450.

McKenzie, D., Z. E. Gedalof, D. L. Peterson, and P. Mote. 2004. Climatic Change, Wildfire, and Conservation. Conservation Biology 18:890-902.

Meigs, G. W., D. C. Donato, J. L. Campbell, J. G. Martin, and B. E. Law. 2009. Forest Fire Impacts on Carbon Uptake, Storage, and Emission: The Role of Burn Severity in the Eastern Cascades, Oregon. Ecosystems 12:1246-1267.

Miller, J. D., H. D. Safford, M. Crimmins, and A. E. Thode. 2009. Quantitative Evidence for Increasing Forest Fire Severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16-32.

Miller, J. D., C. N. Skinner, H. D. Safford, E. E. Knapp, and C. M. Ramirez. 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications 22:184-203.

Moeur, M., T. A. Spies, M. Hemstrom, J. R. Martin, J. Alegria, J. Browning, J. Cissel, W. B. Cohen, T. E. Demeo, S. Healey, and R. Warbington. 2005. Northwest Forest Plan: The first 10 years (1994-2003): status and trend of late-successional and old-growth forest. Gen. Tech. Rep. GTR-PNW-646, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Mote, P. W. and E. P. Salathé. 2010. Future climate in the Pacific Northwest. Climatic Change 102:29-50.

Murphy, E. C. and W. A. Lehnhausen. 1998. Density and foraging ecology of woodpeckers following a stand-replacement fire. Journal of Wildlife Management 62:1359-1372.

Naficy, C., A. Sala, E. G. Keeling, J. Graham, and T. H. DeLuca. 2010. Interactive effects of historical logging and fire exclusion on ponderosa pine forest structure in the northern Rockies. Ecological Applications 20:1851-1864.

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262:703-717.

Perry, D. A., H. Jing, A. Youngblood, and D. R. Oetter. 2004. Forest structure and fire susceptibility in volcanic landscapes of the eastern high cascades, Oregon. Conservation Biology 18:913-926.

PRISM. PRISM Climate Group. http://www.prism.oregonstate.edu/. Accessed April 11 2011.

Rehfeldt, G. E., N. L. Crookston, M. V. Warwell, and J. S. Evans. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences 167:1123-1150.

Rogers, B. M., R. P. Neilson, R. Drapek, J. M. Lenihan, J. R. Wells, D. Bachelet, and B. E. Law. 2011. Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. Journal of Geophysical Research-Biogeosciences 116.

Rothermel, R. 1972. A mathematical model for predicting fire spread in wildland fuels. Res. Pap. INT-115, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Savage, M. and J. N. Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 35:967-977.

Simpson, M. 2007. Forested Plant Associations of the Oregon East Cascades. R6-NR-ECOL-TP-03-2007, Pacific Northwest Research Station, Bend, OR.

Spies, T. A., M. A. Hemstrom, A. Youngblood, and S. Hummel. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology 20:351-362.

Stafford, S. G., D. M. Bartels, S. Begay-Campbell, J. L. Bubier, J. C. Crittenden, S. L. Cutter, J. R. Delaney, T. E. Jordan, A. C. Kay, G. D. Libecap, J. C. Moore, N. N. Rabalais, D. Rejeski, O. E. Sala, J. M. Shepherd, and J. Travis. 2010. Now is the time for action: Transitions and tipping points in complex environmental systems. Environment 52:38-45.

Thompson, C. M., W. J. Zielinski, and K. L. Purcell. 2011. Evaluating Management Risks Using Landscape Trajectory Analysis: A Case Study of California Fisher. Journal of Wildlife Management 75:1164-1176.

Thorne, J. H., B. J. Morgan, and J. A. Kennedy. 2008. Vegetation change in the central Sierra Nevada, California, USA. Madroño 55:223-237.

USFS. 2004. Davis Fire recovery project final environmental impact statement. Deschutes National Forest, Oregon.

USFS. 2005. B & B Fire recovery project final environmental impact statement. Sisters Ranger District, Deschutes National Forest, Oregon.

USFS. Deschutes & Ochoco National Forests & Crooked River National Grassland. http://www.fs.fed.us/projects/hfi/2003/august/documents/deschutes-fact-sheet.pdf. Accessed September 2012.

van Wagtendonk, J. W. 2007. The history and evolution of wildland fire use. Fire Ecology 3:3-17.

Waring, R. H., N. C. Coops, and S. W. Running. 2011. Predicting satellite-derived patterns of large-scale disturbances in forests of the Pacific Northwest Region in response to recent climatic variation. Remote Sensing of Environment 115:3554-3566.

Westerling, A. L., A. Gershunov, T. J. Brown, D. R. Cayan, and M. D. Dettinger. 2003. Climate and wildfire in the western United States. Bulletin of the American Meteorological Society 84:595-+.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313:940-943.

Westerling, A. L., M. G. Turner, E. A. H. Smithwick, W. H. Romme, and M. G. Ryan. 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences of the United States of America 108:13165-13170.

Williams, J. W. and S. T. Jackson. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment 5:475-482.

Williams, J. W., S. T. Jackson, and J. E. Kutzbacht. 2007. Projected distributions of novel and disappearing climates by 2100 AD. Proceedings of the National Academy of Sciences of the United States of America 104:5738-5742.

Wimberly, M. C. and R. S. H. Kennedy. 2008. Spatially explicit modeling of mixed-severity fire regimes and landscape dynamics. Forest Ecology and Management 254:511-523.

Wright, C. S. and J. K. Agee. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecological Applications 14:443-459.

Appendix

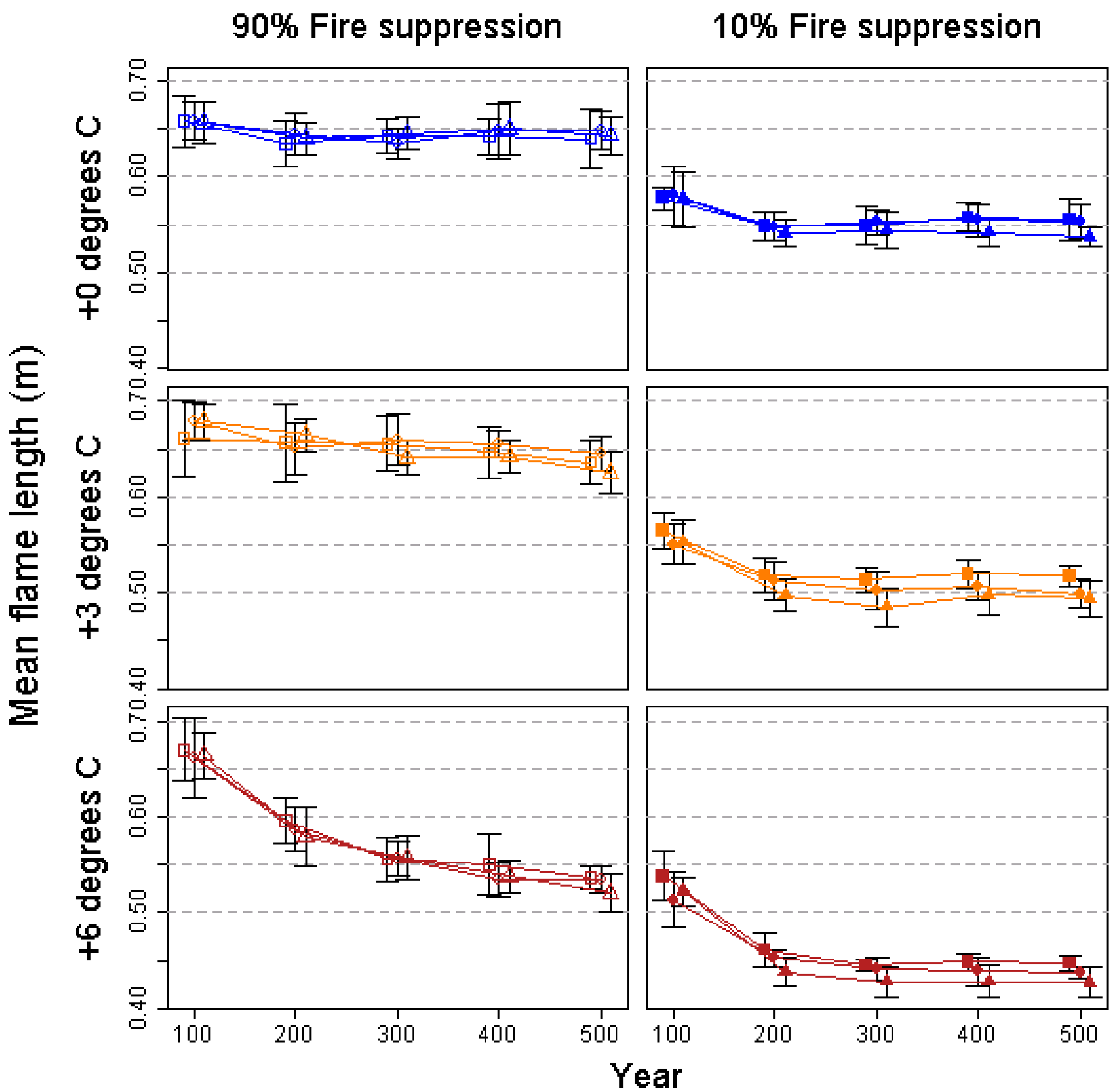

Figure A1. Area-weighted mean flame lengths for each scenario over time. Whiskers are standard deviation. Values represent averages over the prior 100 years. Circles: current precipitation; squares: +10% precipitation; triangles: -10% precipitation.

Table A1. Major FireBGCv2 input data sources.

Parameter	Data description	Source	Availability
Site boundaries	Potential vegetation type	Landscape Ecology Modeling, Mapping & Analysis	http://www.fsl.orst.edu/lemma/
Soils	Soil depth, soil fractional components	Deschutes National Forest	http://www.fs.fed.us/r6/data-library/gis/deschutes/index.shtml
Soils	Soil depth, soil fractional components	Willamette National Forest	http://www.fs.fed.us/r6/data-library/gis/willamette/index.shtml
Daily weather	Historical weather stream (1942-2010)	Western Regional Climate Center: Wickiup Dam	http://www.wrcc.dri.edu/
Weather	Interpolation of missing weather data	DAYMET	http://daymet.ornl.gov/singlepixel
Weather	Spatial interpolation of weather stream	Numerical Terradynamic Simulation Group: MTCLIM	http://www.ntsg.umt.edu/project/mtclim
Weather	Precipitation isohyets; estimates of adiabatic lapse rates	SNOTEL: Irish Taylor	http://www.wcc.nrcs.usda.gov
Current vegetation	Modeled spatial current vegetation	Landscape Ecology Modeling, Mapping & Analysis	http://www.fsl.orst.edu/lemma/
Current vegetation (non-forest)	Current vegetation for meadows, shrubland	FERA: Fuel Characteristic Classification System	http://www.fs.fed.us/pnw/fera/fccs/
Species attributes (major source)	Species morphology, phenology, & physiology	(Burns and Honkala 1990)	http://www.na.fs.fed.us/spfo/pubs/silvics_manual/table_of_contents.htm
Species attributes (major source)	Species morphology, phenology, & physiology	Fire Effects Information System	http://www.fs.fed.us/database/feis/
Fuel loadings	1-, 10-, 100-, & 1000-hour fuel loadings for each vegetation type	FERA: Fuel Characteristic Classification System	http://www.fs.fed.us/pnw/fera/fccs/
Historical fire	Modeled spatial estimate of historical fire return intervals	LANDFIRE	http://www.landfire.gov/

Table A2. Major FireBGCv2 site parameters. PIPO = ponderosa pine; PICO = lodgepole pine; MC = Mixed conifer; TSME = mountain hemlock; ABAM = silver fir.

	Site name	Analysis area (ha)	Elevation (m)	Maximum leaf area index (all-sided)	Maximum stand basal area (m2/ha)	Maximum sapling density (trees/m2)	Average fire frequency (years)	Average fire size (ha)
1	low-mid-rock/developed	133	1425	0.5	1	0.00	1000	100
2	low-dry-PIPO/PICO	8826	1380	7.0	50	0.11	50	80
3	low-dry-PICO	2160	1388	6.0	50	0.10	160	85
4	low-dry-MC	6845	1441	6.5	50	0.10	60	70
5	mid-dry-MC	8547	1547	7.0	55	0.10	62	100
6	mid-dry-MC	234	1475	7.0	60	0.09	65	85
7	mid-mid-MC/TSME	9378	1578	10.0	65	0.16	450	50
8	high-mid-MC/TSME	3739	1818	12.0	65	0.16	550	30
9	high-wet-TSME/ABAM	1188	1713	10.0	52	0.15	550	30
10	peak-mid-TSME	193	1962	10.0	65	0.11	550	30
11	peak-wet-TSME	445	2185	10.0	55	0.04	500	30

Table A3. Scientific and common names of simulated species, with four-letter codes.

Scientific name	Common name	Code
Abies amabilis	Silver fir	ABAM
Abies grandis/concolor	Grand fir/white fir	ABGR
Abies lasiocarpa	Subalpine fir	ABLA
Abies procera/magnifica shastensis	Noble fir/Shasta red fir	ABPR
Calocedrus decurrens	Incense-cedar	CADE
Chrysolepis chrysophylla	Giant chinquapin	CHCH
Cornus nuttallii	Pacific dogwood	CONU
Picea engelmannii	Engelmann spruce	PIEN
Pinus albicaulis	Whitebark pine	PIAL
Pinus contorta	Lodgepole pine	PICO
Pinus lambertiana	Sugar pine	PILA
Pinus monticola	Western white pine	PIMO
Pinus ponderosa	Ponderosa pine	PIPO
Pseudotsuga menziesii	Douglas-fir	PSME
Taxus brevifolia	Western yew	TABR
Tsuga heterophylla	Western hemlock	TSHE
Tsuga mertensiana	Mountain hemlock	TSME

Table A4. Simulation carbon results. Standard deviation in parentheses.

		Mean net primary production (kg/m^2)	Mean total landscape carbon (kg/m^2)
		90% Fire suppression	
	Dry	0.07 (0.0009)	22.1 (0.21)
+0°	*No change*	0.073 (0.0014)	22.4 (0.34)
	Wet	0.074 (0.0008)	22.7 (0.16)
	Dry	0.07 (0.0011)	21.6 (0.35)
+3°	*No change*	0.072 (0.0014)	22 (0.46)
	Wet	0.076 (0.0013)	22.3 (0.44)
	Dry	0.043 (0.0009)	15 (0.43)
+6°	*No change*	0.044 (0.0005)	15.2 (0.45)
	Wet	0.046 (0.0008)	15.5 (0.42)
		10% Fire suppression	
	Dry	0.075 (0.001)	19.9 (0.18)
+0°	*No change*	0.077 (0.0013)	20.5 (0.38)
	Wet	0.081 (0.0013)	20.8 (0.18)
	Dry	0.07 (0.0017)	17.7 (0.74)
+3°	*No change*	0.073 (0.0012)	18.2 (0.56)
	Wet	0.076 (0.0012)	18.9 (0.39)
	Dry	0.04 (0.0011)	11.1 (0.55)
+6°	*No change*	0.042 (0.0009)	11.4 (0.37)
	Wet	0.044 (0.0008)	11.8 (0.44)

Table A5. Crosswalk for assigning general forest type from species importance value forest type. See Table A3 for species codes.

Importance value type	Consolidated type	Importance value type	Consolidated type	Importance value type	Consolidated type
ABAM	Cool wet conifer	ABGR-PIEN	Warm moist conifer	ABLA-TSME	Cool wet conifer
ABAM-ABGR	Cool wet conifer	ABGR-PILA	Warm dry conifer	ABPR	Cool wet conifer
ABAM-ABLA	Cool wet conifer	ABGR-PIMO	Warm moist conifer	ABPR-ABAM	Cool wet conifer
ABAM-ABPR	Cool wet conifer	ABGR-PIPO	Warm dry conifer	ABPR-ABGR	Cool wet conifer
ABAM-CADE	Cool wet conifer	ABGR-PSME	Warm dry conifer	ABPR-ABLA	Cool wet conifer
ABAM-MIX	Cool wet conifer	ABGR-TABR	Warm moist conifer	ABPR-CADE	Cool wet conifer
ABAM-PIAL	Cool wet conifer	ABGR-TSHE	Warm moist conifer	ABPR-CHCH	Cool wet conifer
ABAM-PICO	Cool wet conifer	ABGR-TSME	Warm moist conifer	ABPR-CONU	Cool wet conifer
ABAM-PIEN	Cool wet conifer	ABLA	Cool wet conifer	ABPR-MIX	Cool wet conifer
ABAM-PIMO	Cool wet conifer	ABLA-ABAM	Cool wet conifer	ABPR-PIAL	Cool wet conifer
ABAM-PIPO	Cool wet conifer	ABLA-ABGR	Cool wet conifer	ABPR-PICO	Cool wet conifer
ABAM-PSME	Cool wet conifer	ABLA-ABPR	Cool wet conifer	ABPR-PIEN	Cool wet conifer
ABAM-TSHE	Cool wet conifer	ABLA-CADE	Cool wet conifer	ABPR-PILA	Cool wet conifer
ABAM-TSME	Cool wet conifer	ABLA-CHCH	Cool wet conifer	ABPR-PIMO	Cool wet conifer
ABGR	Warm moist conifer	ABLA-CONU	Cool wet conifer	ABPR-PIPO	Cool wet conifer
ABGR-ABAM	Cool wet conifer	ABLA-MIX	Cool wet conifer	ABPR-PSME	Cool wet conifer
ABGR-ABLA	Cool wet conifer	ABLA-PIAL	Cool wet conifer	ABPR-TABR	Cool wet conifer
ABGR-ABPR	Cool wet conifer	ABLA-PICO	Cool wet conifer	ABPR-TSHE	Cool wet conifer
ABGR-CADE	Warm dry conifer	ABLA-PIEN	Cool wet conifer	ABPR-TSME	Cool wet conifer
ABGR-CHCH	Warm moist conifer	ABLA-PILA	Cool wet conifer	CADE	Warm dry conifer
ABGR-CONU	Warm moist conifer	ABLA-PIMO	Cool wet conifer	CADE-ABAM	Cool wet conifer
ABGR-MIX	Warm moist conifer	ABLA-PIPO	Cool wet conifer	CADE-ABGR	Warm dry conifer
ABGR-PIAL	Warm moist conifer	ABLA-PSME	Cool wet conifer	CADE-ABLA	Warm moist conifer
ABGR-PICO	Warm dry conifer	ABLA-TABR	Cool wet conifer	CADE-ABPR	Warm moist conifer

Table A5 (Continued).

Importance value type	Consolidated type	Importance value type	Consolidated type	Importance value type	Consolidated type
CADE-CONU	Warm dry conifer	CHCH-PSME	Warm dry conifer	PIAL-ABLA	Cool wet conifer
CADE-MIX	Warm dry conifer	CHCH-TABR	Warm moist conifer	PIAL-ABPR	Cool wet conifer
CADE-PIAL	Warm dry conifer	CHCH-TSHE	Warm moist conifer	PIAL-CADE	Warm dry conifer
CADE-PICO	Warm dry conifer	CHCH-TSME	TSME	PIAL-CHCH	Warm dry conifer
CADE-PIEN	Warm moist conifer	CONU	Warm moist conifer	PIAL-CONU	Warm moist conifer
CADE-PILA	Warm dry conifer	CONU-ABGR	Warm moist conifer	PIAL-MIX	PICO
CADE-PIMO	Warm dry conifer	CONU-ABLA	Cool wet conifer	PIAL-PICO	PICO
CADE-PIPO	Warm dry conifer	CONU-ABPR	Cool wet conifer	PIAL-PIEN	Cool wet conifer
CADE-PSME	Warm dry conifer	CONU-CADE	Warm dry conifer	PIAL-PILA	Warm dry conifer
CADE-TABR	Warm moist conifer	CONU-CHCH	Shrubs	PIAL-PIMO	Warm moist conifer
CADE-TSHE	Warm moist conifer	CONU-MIX	Warm moist conifer	PIAL-PIPO	Warm dry conifer
CADE-TSME	Cool wet conifer	CONU-PIAL	Warm dry conifer	PIAL-PSME	Warm dry conifer
CHCH	Warm dry conifer	CONU-PICO	PICO	PIAL-TABR	Warm moist conifer
CHCH-ABGR	Warm dry conifer	CONU-PIEN	Warm moist conifer	PIAL-TSHE	Warm moist conifer
CHCH-ABLA	Cool wet conifer	CONU-PILA	Warm dry conifer	PIAL-TSME	Warm moist conifer
CHCH-ABPR	Cool wet conifer	CONU-PIMO	Warm moist conifer	PICO	PICO
CHCH-CADE	Warm dry conifer	CONU-PIPO	PIPO	PICO-ABAM	Cool wet conifer
CHCH-CONU	Shrubs	CONU-PSME	Warm dry conifer	PICO-ABGR	Warm dry conifer
CHCH-MIX	Warm moist conifer	CONU-TABR	Warm moist conifer	PICO-ABLA	Cool wet conifer
CHCH-PIAL	Cool wet conifer	CONU-TSHE	Warm moist conifer	PICO-ABPR	Cool wet conifer
CHCH-PICO	PICO	CONU-TSME	TSME	PICO-CADE	Warm dry conifer
CHCH-PIEN	Warm moist conifer	Non-forest	Non-forest	PICO-CHCH	PICO
CHCH-PILA	Warm dry conifer	PIAL	Cool wet conifer	PICO-CONU	PICO
CHCH-PIMO	Warm moist conifer	PIAL-ABAM	Cool wet conifer	PICO-MIX	PICO

Table A5 (Continued).

Importance value type	Consolidated type	Importance value type	Consolidated type	Importance value type	Consolidated type
PICO-PIEN	Cool wet conifer	PIEN-TSME	Cool wet conifer	PIMO-CONU	Warm moist conifer
PICO-PILA	Warm dry conifer	PILA	Warm dry conifer	PIMO-MIX	Warm moist conifer
PICO-PIMO	Warm moist conifer	PILA-ABGR	Warm dry conifer	PIMO-PIAL	Warm moist conifer
PICO-PIPO	PIPO-PICO	PILA-ABLA	Warm dry conifer	PIMO-PICO	Warm moist conifer
PICO-PSME	Warm dry conifer	PILA-ABPR	Warm dry conifer	PIMO-PIEN	Warm moist conifer
PICO-TABR	Warm moist conifer	PILA-CADE	Warm dry conifer	PIMO-PILA	Warm dry conifer
PICO-TSHE	Warm moist conifer	PILA-CHCH	Warm dry conifer	PIMO-PIPO	Warm dry conifer
PICO-TSME	PICO	PILA-CONU	Warm dry conifer	PIMO-PSME	Warm dry conifer
PIEN	Warm moist conifer	PILA-MIX	Warm dry conifer	PIMO-TABR	Warm moist conifer
PIEN-ABAM	Cool wet conifer	PILA-PIAL	Warm dry conifer	PIMO-TSHE	Warm moist conifer
PIEN-ABGR	Warm moist conifer	PILA-PICO	Warm dry conifer	PIMO-TSME	Cool wet conifer
PIEN-ABLA	Cool wet conifer	PILA-PIEN	Warm dry conifer	PIPO	PIPO
PIEN-ABPR	Cool wet conifer	PILA-PIMO	Warm dry conifer	PIPO-ABAM	Warm dry conifer
PIEN-CADE	Warm moist conifer	PILA-PIPO	Warm dry conifer	PIPO-ABGR	Warm dry conifer
PIEN-CHCH	Warm moist conifer	PILA-PSME	Warm dry conifer	PIPO-ABLA	Warm dry conifer
PIEN-CONU	Warm moist conifer	PILA-TABR	Warm dry conifer	PIPO-ABPR	Warm dry conifer
PIEN-MIX	Warm moist conifer	PILA-TSHE	Warm dry conifer	PIPO-CADE	Warm dry conifer
PIEN-PIAL	Warm moist conifer	PILA-TSME	Warm moist conifer	PIPO-CHCH	PIPO
PIEN-PICO	PICO	PIMO	Warm moist conifer	PIPO-CONU	PIPO
PIEN-PILA	Warm dry conifer	PIMO-ABAM	Cool wet conifer	PIPO-MIX	PIPO
PIEN-PIMO	Warm moist conifer	PIMO-ABGR	Warm moist conifer	PIPO-PIAL	Warm dry conifer
PIEN-PIPO	Warm dry conifer	PIMO-ABLA	Cool wet conifer	PIPO-PICO	PIPO-PICO
PIEN-PSME	Warm dry conifer	PIMO-ABPR	Cool wet conifer	PIPO-PIEN	Warm dry conifer
PIEN-TABR	Warm moist conifer	PIMO-CADE	Warm moist conifer	PIPO-PILA	Warm dry conifer

Table A5 (Continued).

Importance value type	Consolidated type	Importance value type	Consolidated type	Importance value type	Consolidated type
PIPO-PSME	Warm dry conifer	TABR-ABLA	Warm moist conifer	TSHE-PICO	Warm moist conifer
PIPO-TABR	Warm dry conifer	TABR-ABPR	Warm moist conifer	TSHE-PIEN	Warm moist conifer
PIPO-TSHE	Warm dry conifer	TABR-CADE	Warm moist conifer	TSHE-PILA	Warm moist conifer
PIPO-TSME	Warm moist conifer	TABR-CHCH	Warm moist conifer	TSHE-PIMO	Warm moist conifer
PSME	Warm dry conifer	TABR-CONU	Warm moist conifer	TSHE-PIPO	Warm moist conifer
PSME-ABAM	Warm moist conifer	TABR-MIX	Warm moist conifer	TSHE-PSME	Warm moist conifer
PSME-ABGR	Warm moist conifer	TABR-PIAL	Warm moist conifer	TSHE-TABR	Warm moist conifer
PSME-ABLA	Warm moist conifer	TABR-PICO	Warm moist conifer	TSHE-TSME	Warm moist conifer
PSME-ABPR	Warm moist conifer	TABR-PIEN	Warm moist conifer	TSME	TSME
PSME-CADE	Warm dry conifer	TABR-PILA	Warm dry conifer	TSME-ABAM	Cool wet conifer
PSME-CHCH	Warm dry conifer	TABR-PIMO	Warm moist conifer	TSME-ABGR	Cool wet conifer
PSME-CONU	Warm dry conifer	TABR-PIPO	Warm dry conifer	TSME-ABLA	Cool wet conifer
PSME-MIX	Warm dry conifer	TABR-PSME	Warm dry conifer	TSME-ABPR	Cool wet conifer
PSME-PIAL	Warm moist conifer	TABR-TSHE	Warm moist conifer	TSME-CADE	Cool wet conifer
PSME-PICO	Warm dry conifer	TABR-TSME	Cool wet conifer	TSME-CHCH	TSME
PSME-PIEN	Warm moist conifer	TSHE	Warm moist conifer	TSME-CONU	TSME
PSME-PILA	Warm dry conifer	TSHE-ABAM	Warm moist conifer	TSME-MIX	TSME
PSME-PIMO	Warm moist conifer	TSHE-ABGR	Warm moist conifer	TSME-PIAL	Cool wet conifer
PSME-PIPO	Warm dry conifer	TSHE-ABLA	Warm moist conifer	TSME-PICO	Cool wet conifer
PSME-TABR	Warm moist conifer	TSHE-ABPR	Warm moist conifer	TSME-PIEN	Cool wet conifer
PSME-TSHE	Warm moist conifer	TSHE-CADE	Warm moist conifer	TSME-PILA	Cool wet conifer
PSME-TSME	Warm moist conifer	TSHE-CHCH	Warm moist conifer	TSME-PIMO	Cool wet conifer
Shrubs	Shrubs	TSHE-CONU	Warm moist conifer	TSME-PIPO	Cool wet conifer
TABR	Warm moist conifer	TSHE-MIX	Warm moist conifer	TSME-PSME	Cool wet conifer